Life Cycle Assessment Of The Production Of Edible Oil Emulsions

Comparing A Novel Process Route Using Aqueously Extracted Oil-Bodies Against Existing Technology

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LIFE CYCLE ASSESSMENT OF THE PRODUCTION OF EDIBLE OIL EMULSIONS -

COMPARING A NOVEL PROCESS ROUTE USING AQUEOUSLY EXTRACTED OIL-BODIES AGAINST EXISTING TECHNOLOGY

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ABSTRACT

It is estimated that over a third of the diet in the Western world is made up of oils and fats, of which a prominent percentage is in the form of emulsion food products, including milks, creams, yoghurts, margarines, salad dressings, desserts, soups and cheese. Current processing techniques involve the extraction and refining of edible oils using high temperatures and organic solvents, followed by re-encapsulation of the oil, for incorporation into the required emulsion products.

The research presented in this PhD thesis was performed within the auspices of the UK Department of Environment, Food and Rural Affairs (DEFRA) funded, Sustainable Emulsion Ingredients through Bio-Innovation (SEIBI) project, which involved collaboration with researchers from the University of Nottingham together with a consortium of industrial partners. SEIBI was initiated to investigate a novel processing route for the production of food-grade rape and sunflowerseed oil emulsions from aqueously extracted oil-bodies. Being less energy and chemical intensive, the novel process offered potential reductions in both greenhouse gas emissions and wider environmental impacts when compared with conventional processing.

Using Life Cycle Assessment (LCA) techniques, the environmental burdens of the aqueous oil-body extraction process were determined and compared with those of the existing technology route. To facilitate this, the research focussed on six key objectives, designed to both identify the environmental loads of the systems involved and scrutinise the impact of a number of methodological choices for LCA. These included choice of allocation method, normalisation, scaling issues distinct for novel processes and the extent to which the single-issue LCA variant, carbon footprinting could be used as an environmental indicator for the system.

LCAs for four separate categories of product systems were developed encompassing seed oils, mayonnaises, aqueously extracted oil-body materials and mayonnaise-like oil-body emulsions. In addition to generating the environmental profiles required to fulfil the research objectives, the analysis of these models enabled the generation of original knowledge through the quantification of impacts for a range of processes that had either not previously been assessed or for which no published data could be found.

The novel process was concluded as having clear potential for improved environmental performance over current technology even in its' pre-optimised, although the methodological choices examined were found to have profound effects on these and other results. Oil-body yield from seed was identified as key for optimisation to further maximise the environmental gains, with modest improvements, well within those theoretically possible being required for the novel process to better the environmental credentials of current technology in all key impact areas.

The original outputs from this thesis will be of considerable use to developers involved in the continued advancement of the oil-body extraction technology, together with researchers within the edible oils and emulsions sector. In addition, the methodological outputs will help to inform LCA practitioners and developers in the continuing quest to understand the capabilities and limitations of this powerful analytical tool.

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And finally, I could never have accomplished this if it weren't for the support that I have received from my husband Steve and children Will and Katie to whom I owe an enormous debt (….not financial W and K!). I know that they will share my relief at finishing this thesis – although I'm sorry to report that it may take a while for sanity and normality to return (if ever!).

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DEDICATION

To my Mum:

Who had unwavering belief in me & inspired me in everything I do.

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GLOSSARY OF TERMS

ACRONYM	TITLE AND DESCRIPTION
AOCs	American Oil Chemists Society
AoP	Area of protection – endpoint damage category
ALO	Agricultural land occupation
CC	Climate change: HH = Climate change impact on human health (endpoint category) ES = Climate change impact on ecosystems (endpoint category)
CFC	Chloroflourocarbon compound
CFP	Carbon footprint
CML	Institute of Environmental Sciences of Leiden University, The Netherlands. (Centrum Milieukunde Leiden)
CRO	Crude rapeseed oil
DB	Dichlorobenzene
EBRD	European Bank for Reconstruction and Development
ED	Damage to Eco-system Diversity (ReCiPe(2008)) damage category
ES	Damage to Ecosystems - ReCiPe (2008) damage category
FAO	Food and Agriculture organisation of the United Nations
FD	Fossil depletion
FDA	United States Food and Drug Administration *The United States' government regulatory body responsible for protecting the public health by assuring the safety, efficacy and security of human and veterinary products. (www.fda.gov, 2013)*
FE	Freshwater eutrophication
FET	Freshwater eco-toxicity
FEDIOL	The federation representing the European Vegetable Oil and Protein meal Industry in Europe
GHG	Greenhouse gas
GLO	Global
HH	Damage to Human Health - ReCiPe(2008) damage category
HT	Human toxicity
IMF	International Monetary Fund *An organization of 188 countries, working to foster global monetary cooperation, secure financial stability, facilitate international trade,*

	promote high employment and sustainable economic growth, and reduce poverty around the world (www.imf.org, 2013)
IPCC	Intergovernmental Panel on Climate Change *Established by the United Nations Environment Programme (UNEP) and the World Meteorological Organization (WMO) in 1988, the IPCC is the world's leading international body for the assessment of climate change (www.ipcc.ch, 2013)*
IR	Ionising radiation
ISO	International Standards Organisation *Founded in 1947, ISO (International Organization for Standardization) is the world's largest developer of voluntary International Standards (www.iso.org (2013)*
ISO/TR	International Standards Organisation - Technical Report *An informative document containing information of a different kind from that normally published in a normative document.*
ISO/TS	International Standards Organisation - Technical Specification *A normative document representing the technical consensus within an ISO committee*
JRC	European Commission Joint Research Centre
LCA	Life Cycle Assessment
MD	Metal depletion
ME	Marine eutrophication
MET	Marine eco-toxicity
NLT	Natural land transformation
NOx	Oxides of nitrogen
OBM	Oil-body mayonnaise
OD	Ozone depletion
ODS	Ozone depleting substance
OSR	Oilseed rape
O/W	Oil in Water
PAH	Polycyclic aromatic hydrocarbons, also known as Polynuclear Aromatic Hydrocarbons
PCB	Polychlorinated Biphenyls
PMF	Particulate matter formation
POF	Photochemical oxidant formation
RA	Damage to Resource Availability - ReCiPe(2008) damage category

RER	All European countries, including those not in the European Community (EC)
RIVM	Dutch National Institute For Public Health and the Environment
RRO	Refined rapeseed oil
SETAC	Society of Toxicology and Chemistry *A not-for-profit, global professional organization that since 1979, has provided a forum for scientists, managers and other professionals exchange information and ideas on the study, analysis and solution of environmental problems, the management and regulation of natural resources, research and development, and environmental education*. (www.setac.org, 2013)
SFO	Sunflowerseed oil
TA	Terrestrial acidification
TET	Terrestrial eco-toxicity
ULO	Urban land occupation
USDA	United States Department of Agriculture
VOS	Volatile organic substance
WHO	World Health Organisation
WOB	Wet oil-bodies

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CHAPTER 1. INTRODUCTION

"A point has been reached in history when we must shape our actions throughout the world with a more prudent care for their environmental consequences"

Declaration of the United Nations Conference on the Human Environment, Stockholm (1972).

Whilst 'environmentalism' can trace its roots back through centuries rather than decades, it is widely accepted that a progressive increase in general awareness of the environmental issues facing the earth began in the late nineteen sixties, with many proclaiming Rachel Carson's book 'The Silent Spring' as the catalyst to action (Dodds et al., 2012). Steady progress has been made in the decades since, both in analysing and understanding the multitude of environmental challenges facing the world and publicising those issues to an extent where the general public have become familiar with terms such as terms 'Ozone Depletion', 'Acid Rain', 'Chemical Smog' and most recently 'Global Warming' and 'Climate Change'.

It is now widely accepted that in order to support the worlds continually growing population, processes and products need to generate reduced environmental burdens to utilise the earth's resources in a sustainable manner and mitigate environmental degradation. The UK Government set targets to reduce Greenhouse Gas (GHG) emissions by at least 12.5% by 2012 and 80% by 2050 compared with the baseline emissions of 1990 (www.gov.co.uk, 2014) and this has lead to carbon reduction targets being set throughout industry sectors and individual companies to meet those levels.

Food plays a vital role in the life of every living being. On a basic level it is an essential requirement for human existence, providing the necessary fuel (calories) and nutrients for the body to function, but it also provides pleasure through both consumption and the social aspects of eating. A report by the World Health Organisation (WHO, 2005) estimates that presently the global aggregate food production is sufficient to meet the needs of the current population, however as the population continues to rise, so must the production and distribution of food, with the Food and Agriculture Organisation (FAO) of the United Nations predicting that a 70% increase in food production will be required to meet the needs of the population in 2050 (FAO, 2009).

Foster et al. (2006) note that '*there is general agreement that the production, processing, transport and consumption of food accounts for a prominent portion of the environmental burden imposed by any Western European country'*. Furthermore, it is estimated that food as a whole contributes 15 to 28 % of the overall greenhouse gas (GHG) emissions in developed countries, with all stages in the supply chain, from agricultural production through processing, distribution, retailing, home food preparation and waste, playing a part (Garnett, 2013). In the UK, Food and Drink Federation (FDF) members are committed to an industry-wide absolute target to reduce CO_2 emissions by 35% by 2020 against a 1990 baseline measured within their voluntary Climate Change Agreement with the Department for Energy and Climate Change (DECC) (www.fdf.org.uk, 2013).

Life Cycle Assessment (LCA) has been used widely as a tool to quantify the full range of environmental impacts of systems across the supply chain including food products. Anderson and Ohlsson (1998), Foster et al. (2006), Schau and Fet (2008) and Roy et al. (2009) all provide information on the multitude and variety of LCA studies performed in this sector and Notarnicola et al. (2012) highlight how important it is that '*we do more integrated LCA studies with regard to our entire food production and consumption system'*. In recent years however, the increased focus on GHG accounting over the entire supply chain has lead to the popularity of the single-issue LCA variant carbon footprinting (CFP) soaring. Williams et al. (2012) note that carbon footprinting is one of the foremost methods available for helping tackle the threat of climate change through quantifying anthropogenic GHG impact. However, as noted by Finkbeiner (2009) climate change is not the only environmental issue of relevance and therefore carbon footprint (CFP) '*is not in all cases the right proxy to support sustainable production and consumption.*'

Great potential for environmental improvement exists through using LCA techniques at the earliest possible stage for process and product development as it is estimated that about 80% of all environmental effects associated with a product are determined in the design phase of development (Tischner, 2000). Its use in this way does however provide methodological and practical difficulties (Hetherington et al., 2014), for which it is important to generate an awareness and understanding.

1.1. **RESEARCH CONTEXT**

The research outlined throughout this thesis was performed within the auspices of the UK Department of Environment, Food and Rural Affairs (DEFRA) funded, collaborative and cross disciplinary Sustainable Emulsion Ingredients through Bio-Innovation (SEIBI) project involving collaboration with researchers from the University of Nottingham together with a consortium of industrial partners. The SEIBI project was initiated to investigate a novel processing route for the production of rape and sunflowerseed oil emulsions for food production which was perceived to have environmental benefits over current technology. This chapter outlines the key elements that frame the motivation and purpose of the research contained in this thesis, outlining the context to better understand the research aims.

It is estimated that 35-40% of the diet in the Western world is made up of edible oils and fats (www.lipidlibrary.aocs.org, 2013), whether they are in dairy or other animal products, spreads, confectionary items or cooking oils (AOCS, 2011). Current oilseed processing techniques require extraction and refining of the oil using high temperatures and organic solvents, followed by re-encapsulation of the oil if required using surfactants, for incorporation into the required food products. The processes for oil production have remained largely unchanged for decades, with minor changes designed largely to increase mechanical efficiency Dijkstra (2009). The SEIBI project aimed to develop alternative extraction and processing routes for the emulsions, reducing the number and complexity of processing steps required for these processes, with the intention that simplification will improve efficiency and reduce the environmental impact of the production of edible oil emulsions.

An essential part of the project is the determination of the environmental burdens of food grade rape and sunflowerseed oil emulsion production via both traditional and novel process routes, to verify whether the novel route offers environmental benefits and also utilise this information to help development of the early-stage process. This analysis was performed using Life Cycle Assessment (LCA) techniques, to identify both the GHG emissions and wider environmental burdens. It is this element of the project that will be presented and discussed in this thesis.

1.2. **EDIBLE OILS**

Edible oils can be of animal or plant origin, however the oils most often associated with this term are those extracted from cultivated plants, hence they are sometimes referred to as vegetable oils. Bockisch (1998) characterises vegetable oils as seed oils and pulp oils (also known as fruit oils), as shown in figure 1.2-1. Examples of the former include sunflower, rape and linseed oil, with the two most important examples of the latter being olive and palm oil.

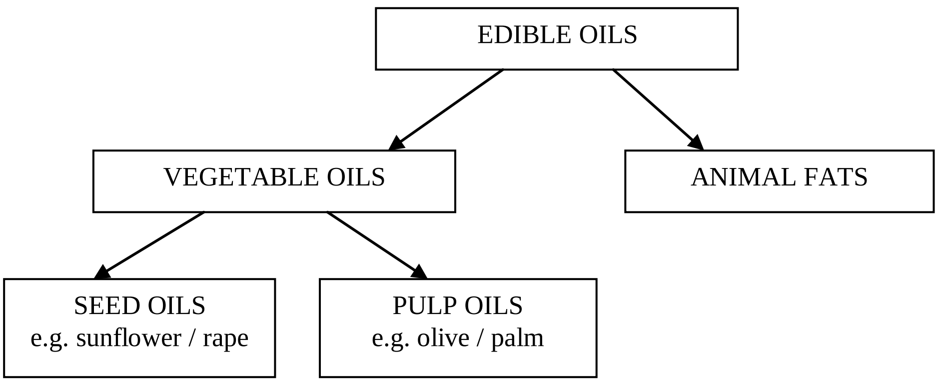

Figure 1.2-1: Types of edible oils as characterised by Bockisch (1998)

There are two major markets for vegetable oils, broadly categorised as food (human and animal feed) and the oleochemical industry, which includes products such as lubricants and bio-fuels. Vegetable oils play an important role within the food system, utilised both commercially and domestically as a processing medium for a variety of foods such as fried products, and as an ingredient in the recipe for a vast array of different foods.

Figures vary for the proportion of the market share utilised for food. Gunstone (2011) reports that until recently it was widely accepted that the ratio for food, animal feed and oleochemical purposes was 80:6:14 respectively and Rosillo-Calle et al. (2009) supports this, citing an 80% global market share for food use. Gunstone (2011a) goes on to note that with the increasing demands from the biodiesel sector this is now better represented by figures such as 75:5:20, whilst data from FEDIOL (the federation representing the European Vegetable Oil and Proteinmeal Industry in Europe) indicates a higher proportion of these oils utilised for non-food use, with only 54% of the vegetable oil produced and imported in the European block of 27 member states (EU-27) used in food production (www.fediol.eu (a)). Table 1.2-1 illustrates the breakdown of uses for vegetable oils within the EU-27 group of countries for this period.

Table 1.2-1: FEDIOL comparison of end-use for all EU-27 vegetable oils in 2010 vs 2011, www.fediol.eu (b)

	Jan – Dec 2010		**Jan – Dec 2011**	
TOTAL	24574 (including 2050 olive oil)		23740 (including 2045 olive oil)	
	Thousand tonnes			
Food	13189	54%	12918	54%
Feed	1000	4%	971	4%
Biodiesel	7636	31%	7680	32%
Non-energy technical	1729	7%	1571	7%
Direct energy (electricity)	800	3%	500	2%
Direct fuel	220	1%	100	0%

Seed oil can be obtained by mechanical pressing (expelling) alone, which extracts up to 90% of the oil, or by a two stage process involving pre-pressing, at a yield of 70-80%, followed by solvent extraction to attain combined yields of up to 99%. For pressing alone, the material must be exposed to higher pressures with longer residence times within the equipment than required for the pre-presses used as part of the combined route.

Blackwell (2010) explains that the difference in capital cost between expelling and the combined route means that solvent extraction is only used by the major processors, where the larger volumes required justify the initial investment to access the higher yields. Since large tonnages are required by the food industry, the solvent extraction route is the most cost-effective in that sector, whilst the lower volume producers such as speciality oil manufacturers favour the simpler method of expelling alone. The process for extraction of seed oils will be outlined in detail in chapter 2 of this thesis.

1.3. OIL-BODIES

Oil-bodies of various types, also known as lipid bodies, are found throughout most if not all plant cells (Murphy, 2001), however seeds are the most common areas where oil-bodies accumulate (ibid). They are the organelles that store the oil within mature seeds (Gray et al., 2010), with true oilseeds accumulating oil-bodies as one of their major storage reserves in amounts ranging from around 20% seed weight in soybean to 42% in rapeseed and as high as 76% in some of the larger seeded nuts Murphy (2001).

Several papers have outlined a technique for aqueous extraction of oil-bodies from oilseeds to generate a natural oil-in-water emulsion (White et al.,2008, Campbell and Glatz, 2009, Nikifordis and Kiosseoglou, 2009, Adams et al., 2012). Aqueous extraction involves the use of water as an extraction medium, into which the soluble cellular materials from the seed dissolve following homogenisation, allowing the release of oil into the bulk liquid phase. The oil can then be recovered from this phase by centrifugation resulting in a natural oil-in-water cream emulsion. Production of emulsions in this way has potential advantages in terms of reduced energy consumption when compared with conventional seed oil processing, together with the removal of a toxic and inflammable organic solvent as part of the process.

Several researchers have highlighted the potential for exploiting this natural oil-in-water emulsion as a more environmentally friendly or sustainable production route for the preparation of food products appearing in the form of emulsions (White et al., 2008, Nikiforidis et al., 2012).

1.4. EDIBLE OIL EMULSIONS AS FOODSTUFFS

A prominent percentage of edible oil production is consumed in the form of emulsion food products. Emulsions consist of two immiscible liquids, with one dispersed in the other in the form of small spherical droplets. McClements (2005) cites common examples of food emulsions as including milk, flavoured milks, creams, whipped cream, butter, yogurt, cheese, salad dressings, mayonnaise, dips, coffee whitener, ice cream, desserts, soups, sauces, margarine, infant formula, and fruit beverages.

Edible oil emulsions exist in two forms (Coupland and McClements, 1996); when oil droplets are dispersed within an aqueous phase, this is known as an oil-in-water (or o/w) emulsion (e.g. mayonnaise, milk, cream and soups), whereas a system that consists of water droplets dispersed in an oil phase is known as a water-in-oil (or w/o) emulsion (e.g. margarine, butter and spreads). Downing (1996) outlines the basic steps for commercial production of an oil-in-water emulsion such as mayonnaise, indicating that the surfactant (egg yolk) is first added to the water and the solution is mixed with an equal volume of oil to form a crude emulsion. This emulsion is then passed through a colloid mill or homogeniser, with more oil being incorporated as required to produce the final product.

1.5. **LIFE CYCLE ASSESSMENT**

In line with the environmental awareness that has grown over the past decades, industry and consumers have turned increasingly to processes and products that are more ‘environmentally friendly’, ‘greener’ or more ‘sustainable’. When making choices or modifying processes to reduce a particular environmental impact however, there is always the danger that the burden may be shifted to a different element of environmental degradation e.g. reduction in the packaging of consumer goods may look to have an entirely positive benefit, but if the result is that more items get damaged before being sold and therefore the level of waste increases, the burden has merely shifted from one area of environmental concern to another. To counteract this, a more holistic approach must be taken, viewing the full range of environmental impacts throughout the entire life cycle of the product or process.

Originally conceived in the late1960’s (Hunt and Franklin, 1996, Astrup Jensen et al., 1997), Life Cycle Assessment (LCA) aims to provide the data to enable the entire environmental profile of products, processes or services to be assessed, thereby providing a more complete indicator of a product’s environmental performance. LCA evaluates the potential environmental impacts of a ‘product system’ by compiling and evaluating a comprehensive inventory of inputs and outputs for that system over its entire life-cycle, i.e. from cradle to grave. ISO14040:2006 defines LCA as the ‘*compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its lifecycle*’.

Whilst a myriad methodological challenges are debated by the LCA community (Ekvall and Weidema 2004; Roy et al. 2009), there is a general consensus on LCA’s suitability as an effective tool for determining environmental credentials (Baumann and Tillman, 2004; Finnveden et al., 2009) with Finkbeiner (2011) noting that it is ‘*the internationally accepted method for measuring environmental performance’*. It has been used extensively to assess the environmental impacts for a variety of systems including product, service and waste management systems in addition to being used widely as a decision-making tool in process selection, design, and optimization (Del Borghi et al. 2007).

Azapagic (1999) noted that acceptance for its use as design and optimisation tool began in the 1990's and whilst Koller et al. (2000) and Tufvesson et al. (2013) note that full-scale LCA is often thought of as too difficult or time consuming to pursue at the research or development stage of a new product or process, determining where improvements can be made whilst a process is still at the laboratory stage can be key to maximising the environmental improvement potential of such a process.

1.6. **CARBON FOOTPRINTING**

In recent years the popularity of the single-issue LCA variant, carbon footprinting (CFP) has soared, with the increased focus on greenhouse gas (GHG) accounting over the entire supply chain fostered by such initiatives as the UK 'Carbon Label' and Sweden's 'Klimatmärkning', borne out of a desire to fulfil GHG reduction commitments. As noted by Laurent et al. (2012), the numerous recent initiatives to standardise CFP and introduce stand-alone GHG accounting methods indicate the level to which they are often the main focus of environmental policies and a search of the bibliographic database 'SCOPUS' on the term 'Carbon Footprint' clearly demonstrates the growth in its usage. There has been an exponential rise in publications on the topic, from fifteen in 2000, to two hundred and ninety five in 2008 when the first CFP standard PAS 2050:2008 was published (revised in 2011), to one thousand and sixty one articles in 2012.

Whereas a full LCA analyses the life cycle of a product system with respect to the entire range of environmental impacts, CFP focuses exclusively on climate change. Whilst such an increase in the popularity of CFP is beneficial for potential GHG reductions, care must be taken to ensure that the narrow scope of assessment does not go against the original rationale for LCA development, and inadvertently shift the burdens to alternative environmental impacts when products are *'optimised to become more "green"'* (Laurent et al., 2012).

1.7. **SCOPE OF THE WORK**

Within the multidisciplinary SEIBI team, researchers at the University of Nottingham have been developing new processing routes for the production of edible oil emulsions from oilseeds, which are less energy intensive and solvent free. The novel process involves the aqueous extraction of oil-bodies from their seeds. As naturally occurring emulsion droplets, the use of oil-bodies removes the need for homogenisation, shear devices or emulsifiers necessary for creating artificial emulsions during conventional production of emulsion food products.

The advantages of this processing route are that the emulsification process step is removed and the extraction of the base material (oil-bodies) is projected to have a lower overall impact on the environment due to the reduced energy intensity of the process and removal of the solvents compared with the extraction and refining stages for the seed oil that it replaces.

In order to assess the environmental credentials of the novel process and provide information to further reduce its environmental impacts whilst still in its early stages, LCA has been used to identify the full range of environmental burdens of the existing processing system together with those that would exist as part of the novel processing route. In addition, an investigation of the suitability of CFP as an environmental performance measure for this system was also carried out to determine whether process improvements based on the CFP results alone would be targeted correctly, or potentially cause burden shifting.

1.8. RESEARCH OBJECTIVES

The primary aim of the work presented in this thesis was to determine whether the novel processing route for production of food grade edible oil emulsions via aqueous extraction of oil-bodies has a better environmental profile than that of the existing technology route.

In order to meet this aim, the research involved the creation of a series of separate, but inter-connected LCA models, focussed on fulfilling six key objectives designed to both identify the environmental loads and scrutinise the impact of number of methodological choices for LCA.

Obj. 1. To use LCA to establish the CFP and wider environmental loads for the production of refined rapeseed and sunflowerseed oils, together with the relative contributions from each of the processing stages.

Whilst many LCAs have been published using seed oils as part of the LCA for a wider product system, only four were identified outlining the environmental performance of rape and sunflowerseed oils as products, (McManus et al., 2004; Narayanaswamy et al., 2004; Schmidt, 2010; Roïz and Paquot, 2013) each of which used different variables within the analysis. In addition to being an essential building block to fulfilling the aim of the research, the output from this LCA will therefore provide valuable information for researchers in the field, through reporting results using a set of methodological choices that are different to those used in previously reports.

Objective 1 will also generate original knowledge in the following areas:

a. Quantification of the impact of using different methods for treatment of co-products within the attributional LCA models

b. Identification of the effect of using different normalisation data sets for analysis of significance.

Obj. 2. To use LCA to identify the CFP and wider environmental loads of the current processing route for production of the case-study food grade emulsion, mayonnaise. No published data exists in this area and these analyses will therefore generate original knowledge through:

a. Identification of the environmental profile and carbon footprint of commercially produced mayonnaise using rape and sunflowerseed oils

b. Evaluation of the relative contribution from each processing stage to the environmental burdens.

Obj. 3. To quantify the CFP and wider environmental impacts of aqueous extraction of oil-bodies from rapeseed and sunflowerseeds. This type of analysis has not previously been performed and will therefore generate the following novel outputs, in addition to progressing towards the principal aim;

a. Identification of the carbon footprint and wider environmental profile for aqueous extraction of oil-bodies from rape and sunflowerseeds

b. Analysis of the assumptions and simplifications required to generate an LCA for the projected commercial scale application of a novel technology that is still at the lab-scale.

Obj. 4. To identify the environmental loads of the production of a ‘mayonnaise-like’ emulsion using rape and sunflowerseed oil-bodies. As with objective 3, this type of analysis has not previously been performed and will therefore generate the following novel output:

a. Identification of the environmental profile and CFP of a proposed commercial scale mayonnaise-like emulsion, produced using rape and sunflowerseed oil-bodies

b. Comparison of the relative environmental performance of a product made via the novel process with one made using conventional technology.

Obj. 5. To determine whether the focus on climate change impacts through using the carbon footprint as an indicator of environmental performance would lead to burden shifting when optimising the systems analysed. No analysis of this type has previously been conducted for these systems and the output from this objective will therefore provide the following original contribution.

a. Analysis of the extent to which the carbon footprint and LCA results provide consistent data to enable correct targeting of process improvements for both mature processes and those at the early development stage.

Obj. 6. To produce information to appropriately direct the development of the process routes by the University of Nottingham researchers and enable progression to scale-up of the process. As a process that is in the early stages of development, the output from this objective will provide novel information:

a. To identify the key areas for process optimisation of the oil-body extraction process to further improve environmental credentials.

1.9. THESIS STRUCTURE

This thesis is divided into ten chapters, within which this introductory chapter sets the scene by outlining the aims and objectives of the research in addition to the context of the study. Chapters 2 and 3 go on to provide background information on the systems to be researched, together with an introduction to the tool used for the analysis, LCA, with Chapter 4 then outlining the methodology used for each of the four case studies modelled.

Chapters 5 to 8 contain methodology and analysis specific to each of the four LCA studies that fulfil the six objectives. These cover production of rape and sunflowerseed oils, mayonnaise, aqueous oil-body extraction and a mayonnaise-like oil-body emulsion. Each of the case-study chapters contains specific discussion and conclusions, with chapter 9 containing a summary discussion following the work as a whole, to finalise the analysis and facilitate the drawing of conclusions.

The thesis will culminate in chapter 10, which contains the final conclusions, together with a summary of the original contributions, a brief commentary on the issues and limitations and recommendations for further work.

The thesis structure is illustrated in figure 1.9-1.

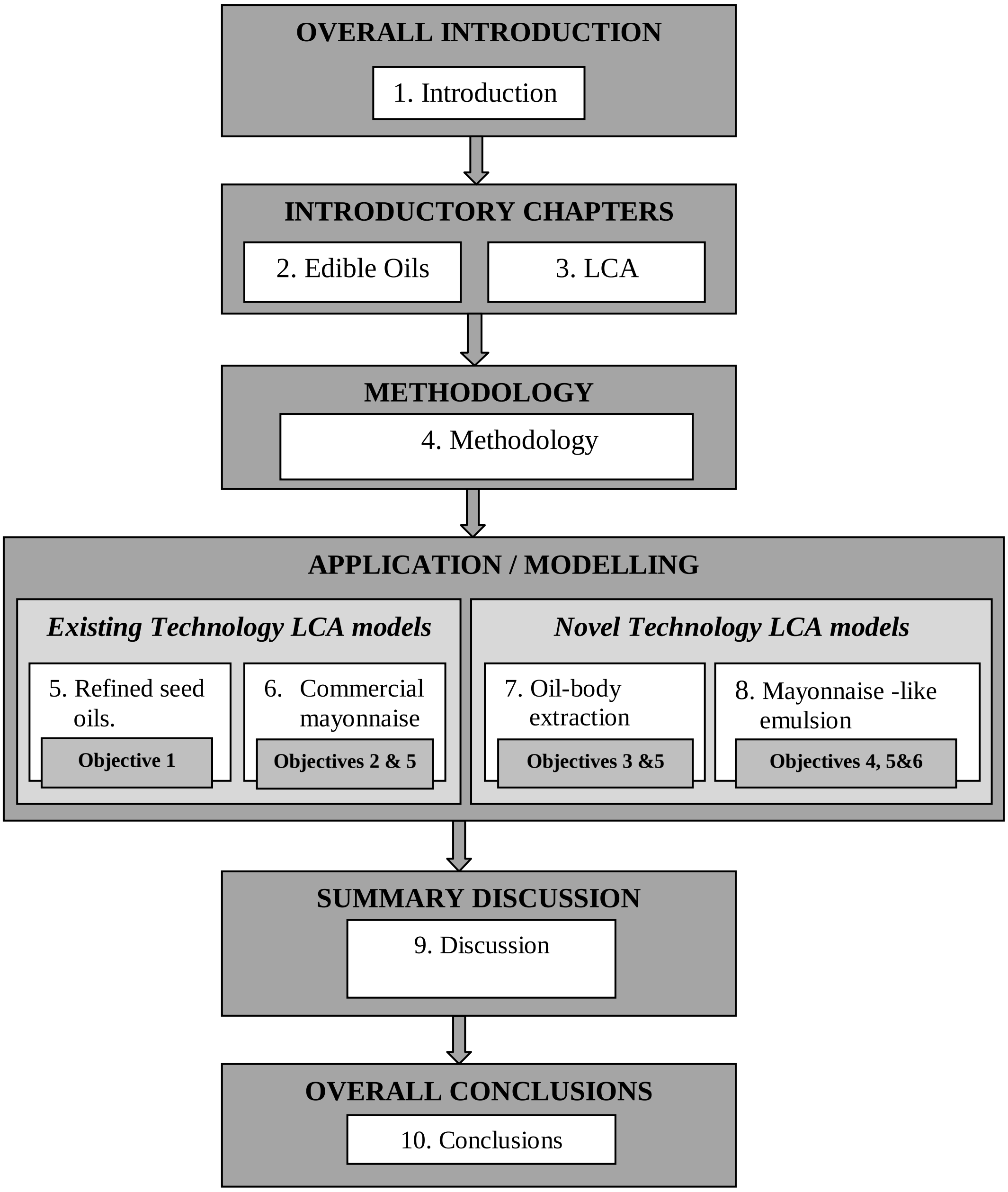

Figure 1.9-1: Thesis Structure

1.10. PUBLICATIONS

The author's work has been disseminated to date through oral presentation of papers at conferences and publication within journals. Details of all such papers are outlined in table 1.10-1 and reproduced in full within appendix A.

Table 1.10-1: Details of publications to date

Citation	Title of paper	Details reproduced in appendix A.
Hetherington et al. (2014).	Does Carbon Footprinting Paint The Right Picture For Process Improvements? A Case Study of Mayonnaise Production**.**	Full paper, currently under review. *Journal of Cleaner Production*. Based on work in chapter 6.
Hetherington et al. (2014).	Use of LCA as a development tool within early research: challenges and issues across different sectors.	Full paper. *International Journal of Life Cycle Assessment.* Vol 19 (1) 130-143 Using material from chapter 7 together with case-study material from co-authors.
Hetherington et al. (2012).	Carbon Footprint Analysis and Life Cycle Assessment of Mayonnaise production. A comparison of their results and messages:	Abstract for paper presented orally. *SETAC Europe 18th LCA Case Study Symposium, 4th NorLCA Symposium, November 26-28 2012, Copenhagen, Denmark.* Based on work in chapter 6.
Hetherington et al. (2011).	Comparison of allocation and impact assessment methodologies on the life cycle assessment of rape and sunflowerseed oils:	Full paper. Presented orally. *LCM 2011, August 28-31, 2011, Dahlem Cube, Berlin, Germany*. Based on work from chapter 5.

CHAPTER 2. EDIBLE OILS

Having outlined the research aims and context in the introductory chapter, it is important to gain an understanding of the characteristics of the system being researched. This chapter outlines key background information for edible oils; their characteristics, cultivation and processing, together with an overview of oil-bodies, the current state of research for the novel processing and the use of both substances in foods as investigated within this research.

2.1. **OVERVIEW**

As outlined in Chapter 1, whist edible oils can be of plant or animal origin, it is vegetable oils which are derived from cultivated plants that are most associated with this term, therefore for the remainder of this thesis, the term ‘edible oils’ will be taken to mean vegetable oils. Gunstone (2013a) highlights rape and sunflowerseed oils as two of the four major commodity vegetable oils behind palm and soyabean. Figure 2.1-1 indicates the growth in production of the four major oils since 1994 and whilst it is clear that the largest growth in production has come from palm and soyabean oils, rape and sunflowerseed oils have also experienced steady growth over this period and are forecast to have continued growth as the needs of a growing population rise.

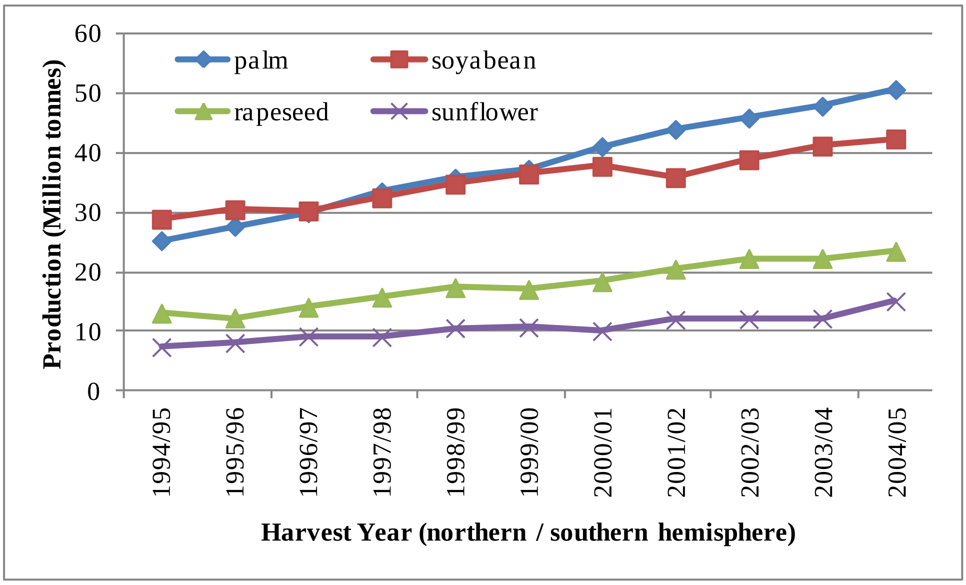

Figure 2.1-1: Production of four major edible oils (million tonnes). Source: Gunstone (2013a)

As discussed in chapter 1, the majority of vegetable oils are consumed as food, representing approximately 75% of the global market, however Gunstone (2013b) notes that in 2011/12 the division of vegetable oils between food and industrial use in Europe was 53% and 47%, respectively, with the breakdown for industrial usage of the individual oils as: rapeseed oil (75%), soybean oil (44%), palm oil (43%), and sunflower oil (6%). In the EU, the 2008/9 Renewable Energy Directive (EC, 2009) set an overall binding target of 20% for the share of EU energy that needs to be met by renewable sources by 2020. As part of this, it is a requirement that10% of the transport fuel for each Member State must come from renewable sources (including biofuels).The impact of this is clearly shown by the increased proportion of edible oils used for non-food purposes in the EU in comparison with global figures, together with the statistic that 75% of rape seed oil within the EU was utilised for industrial purposes in 2011/12 (Gunstone, 2013b).

Whist the lion's share of EU rapeseed oil was utilised for industrial use, 2.3 million tonnes were consumed for food use with consumption of sunflowerseed oil being 3.42 million tonnes (USDA figures in Gunstone, 2013b). In addition to being used as cooking oils, rape and sunflowerseed oil are components in an enormous variety of food products ranging from salad oil, dressings and mayonnaise, to margarines and spreads, chocolate and ice cream fats, bakery fats, confectionery filling and coating fats, vegetable fats for dairy products and fats for infant nutrition (www.fediol.eu(c)).

2.2. CHARACTERISTICS OF EACH SEED

2.2.1. Sunflower

The Sunflower is a composite annual plant, probably originating from the South-West United States-Mexico area, FAO/EBRD (1999). It was introduced into Europe in the 16th century and became established as an oil-seed crop in Eastern Europe. According to Fediol (2011d) Sunflower became very popular as a cultivated plant in the 18th century. Each flower may bear up to 2000 seeds, which depending on variety, may be black, dark brown, grey-brown, beige or striped (ibid). Whilst the majority of the cultivated seeds are utilised to produce oil, the seeds themselves are popular as snacks in Mediterranean and Asian countries, in addition to being used in a variety of health-based snack food products worldwide. The seeds used for seed-oil production are usually smaller than those for snacking (known as confectionary sunflower) which are striped as shown in figure 2.2-1. The seed comprises the outer hull and inner kernel or heart as shown in figure 2.2-2. It is the kernel that is pressed to extract the sunflower oil, as will be outlined later in this chapter.

Figure 2.2-1: Sunflowerseeds: Oil-seeds (left) and confectionary seeds (right). Source: www.fediol.eu (2013

Figure 2.2-2: Left - de-hulled sunflower kernel; Right whole seed. Source: www.en.wikipedia.org (2013)

Sunflowers are mainly grown in warm temperate regions, requiring warm summer months with very dry conditions during the ripening of the seeds. Thus they are not routinely grown in UK, but cultivated in Russia (the biggest individual sunflower producer) Southern and Eastern European countries, Argentina, China and USA. Table 2.2-1 shows the geographical breakdown in million tonnes of sunflower production over the past 6 harvest years.

Table 2.2-1: Production (million tonnes) of sunflowerseed globally.Source: http://lipidlibrary.aocs.org (2013)

	2006/7	2007/8	2008/9	2009/10	2010/11	2011/12
Argentina	3.50	4.65	2.90	2.30	3.67	3.34
Russia	6.75	5.65	7.35	6.42	5.35	9.63
Turkey	0.85	0.70	0.83	0.80	1.00	0.92
Ukraine	5.30	4.20	7.00	7.60	8.40	10.50
EU-27	6.48	4.8	6.94	6.91	6.90	8.29
Other	7.30	7.02	8.01	7.59	8.15	7.61
Total Production	30.18	27.02	33.03	31.62	33.46	40.30
Total Crushed	26.87	24.11	28.65	29.19	29.89	36.71

2.2.2. **Rapeseed**

Oil seed rape (Rapeseed) is a member of the Cruciferae family and grows to a height of 75-175 cm. It has yellow flowers, blue-green leaves and yields seeds that are small, round and black-red in colour as can be seen in figure 2.2-3.

Figure 2.2-3. Rapeseed: seed pod (source:www.commons.wikimedia.org 2013b)

Figure 2.2-4. Rapeseed: seeds alone (source: www.grainscanada.gc.ca, 2013)

Fediol (2011e) states that Rapeseed is one of the oldest cultivated plants on earth, with Gupta and Pratap (2007) noting that the oldest references regarding origin and cultivation of rapeseed come from Asia. Rapeseed oil was originally used mainly for lighting and as a lubricant however it is now one of the most important vegetable oils for present day human consumption, in addition to growing in importance as a source of bio-fuel. Rapeseed varieties were originally rich in erucic acid, however health concerns surrounding dietary exposure to this 22–carbon monounsaturated fatty acid lead to selective breeding to produce low-erucic varieties (FSANZ, 2003). As such the low-erucic acid canola varieties comprise almost the entire rapeseed crop produced in the world today (FSANZ, 2003) and in North America and certain other parts of the world it is more generally know as canola, rather than rape.

Rapeseed is grown particularly in Northern European Countries including UK. Outside Europe the dominant producers are China, India, Canada and Australia. Table 2.2-2 shows the geographical breakdown in million tonnes of sunflower production over the past 6 harvest years. Unlike sunflower, the seeds themselves are not generally utilised as a food (apart from some bird foods) and therefore nearly 99% of the seeds are crushed for oil.

Table 2.2-2: Production (million tonnes) of rape seed globally. Source: http://lipidlibrary.aocs.org (2013)

	2006/7	2007/8	2008/9	2009/10	2010/11	2011/12
China	12.65	10.57	12.10	13.66	13.10	13.43
India	5.80	5.45	7.00	6.40	7.10	6.50
Canada	9.00	9.60	12.64	12.89	12.79	14.61
EU-27	16.01	18.36	19.01	21.55	20.75	19.08
Other	3.34	4.54	7.46	6.46	6.80	7.95
Total Production	46.8	48.52	58.21	60.96	60.55	61.56
Total Crushed	45.03	46.65	52.00	56.55	59.47	60.83

2.3. SEED CULTIVATION

There are two varieties of Rapeseed, spring and winter rape. Schmidt (2007) indicates that winter rape is planted from mid to late August whilst spring rape is planted at the beginning of April. The winter varieties have a longer vegetation period and give a better yield (Fediol, 2011e), but can only be grown in areas with a mild winter climate. In Europe, winter rapeseed is the dominating variety, whereas in Canada only summer rapeseed is grown.

Sunflower is generally planted in March or April and has one of the shortest growing seasons of all the major cash crops in the world (FAO/EBRD, 1999). Early maturing varieties are ready for harvesting ninety to one hundred and twenty days after planting and late maturing varieties take another thirty to forty days.

The cultivation of seed crops such as rape and sunflower broadly encompasses the three stages of 'field preparation', 'sowing' and 'maintenance and harvesting'. These stages are further expanded in figure 2.3-1.

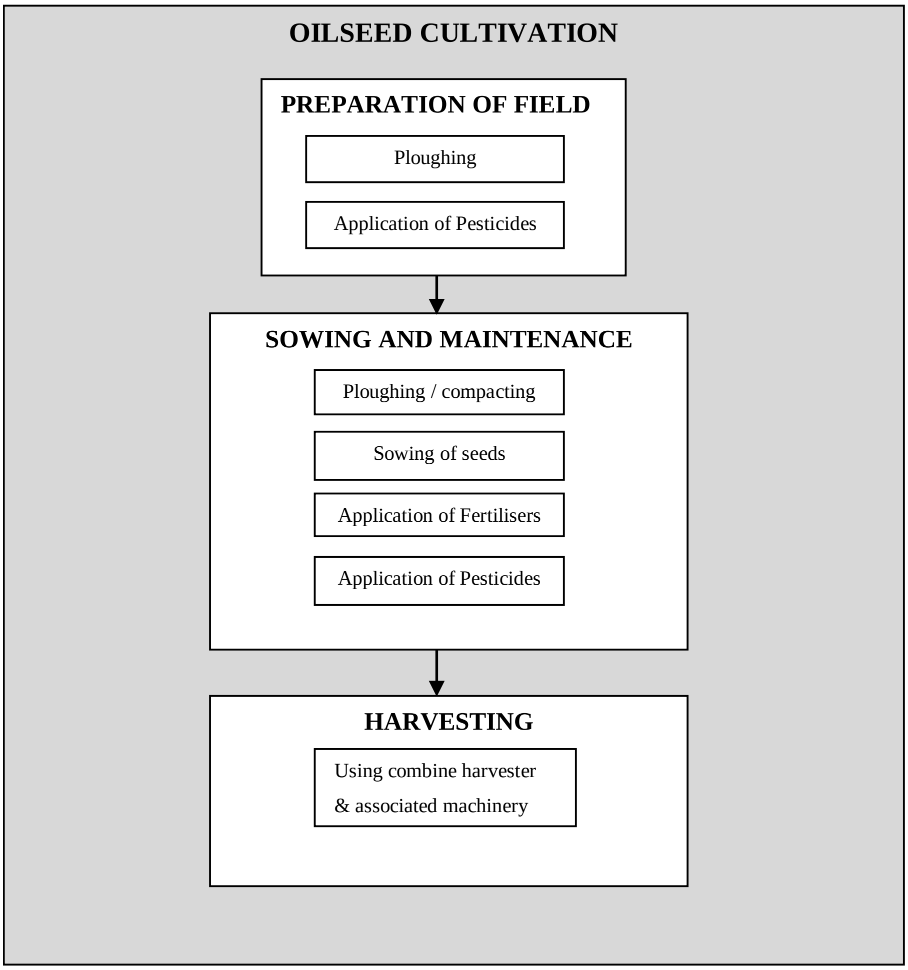

Figure 2.3-1: Schematic overview of oilseed cultivation stages

2.4. OIL SEED PROCESSING

As previously outlined, the production of edible oils can be via mechanical pressing alone, or a two-stage process encompassing pressing followed by solvent extraction. Whilst the capital outlay for a two-stage production facility is high, the large tonnages required by the food industry justify the initial investment to access the higher yields. Thus, whilst lower volume producers such as speciality oil manufacturers favour expelling alone, the predominant method for extracting seed oil for large scale food industry use is primarily physical pressing and solvent extraction (Blackwell, 2010: Van Hoed et al., 2010; Liu et al., 2012). It is this two-stage process which will be utilised for the processes investigated by this research.

The two-stage process broadly comprises three overall stages; seed pre-treatment, pre-pressing and solvent extraction, which together produce the crude seed oil and the associated by-product meal, which is utilised as animal feed. These process stages are as illustrated in figure 2.4-1.

Bockisch (1998) outlines the initial pre-treatment actions, within which the dry seed is conveyed from the storage silo to the screening equipment which incorporates magnetic screening to remove metal contaminants followed by sieving and pneumatic separation of other impurities, such as leaves, twigs or other non-metallic debris.

He goes on to state that most oilseeds that can be extracted after size reduction are also dehulled. Generally the hulls do not contain fat and therefore reduce the capacity of the plant, which Ward (1982) notes causes higher oil-in-meal numbers and increased solvent retention. Matthäus (2007) notes that '*contrary to sunflowerseeds, dehulling of rapeseed is not a common used process, since it is expensive, due to the small size of the seeds*' an observation supported by Bockisch (1998) and Dumelin (2013).

Sunflowerseeds require dehulling as the hulls make up 30% of the seed weight and the removal of the hull also enables the wax content of the oil to be reduced (Bockisch. 1998). They are generally de-hulled by flaking the pre-heated seeds between two smooth surface cast-iron rolls (Ward,1982; McManus et al., 2004; www.whc-oils.com, 2013).

Prior to pressing, both types of seed are thermally treated to complete the rupture of the oil-containing cells, decrease the viscosity of the oil and adjust the natural moisture content of the seed. The resultant meal-cake material is then conveyed to the pre-pressing equipment for the first stage of extraction.

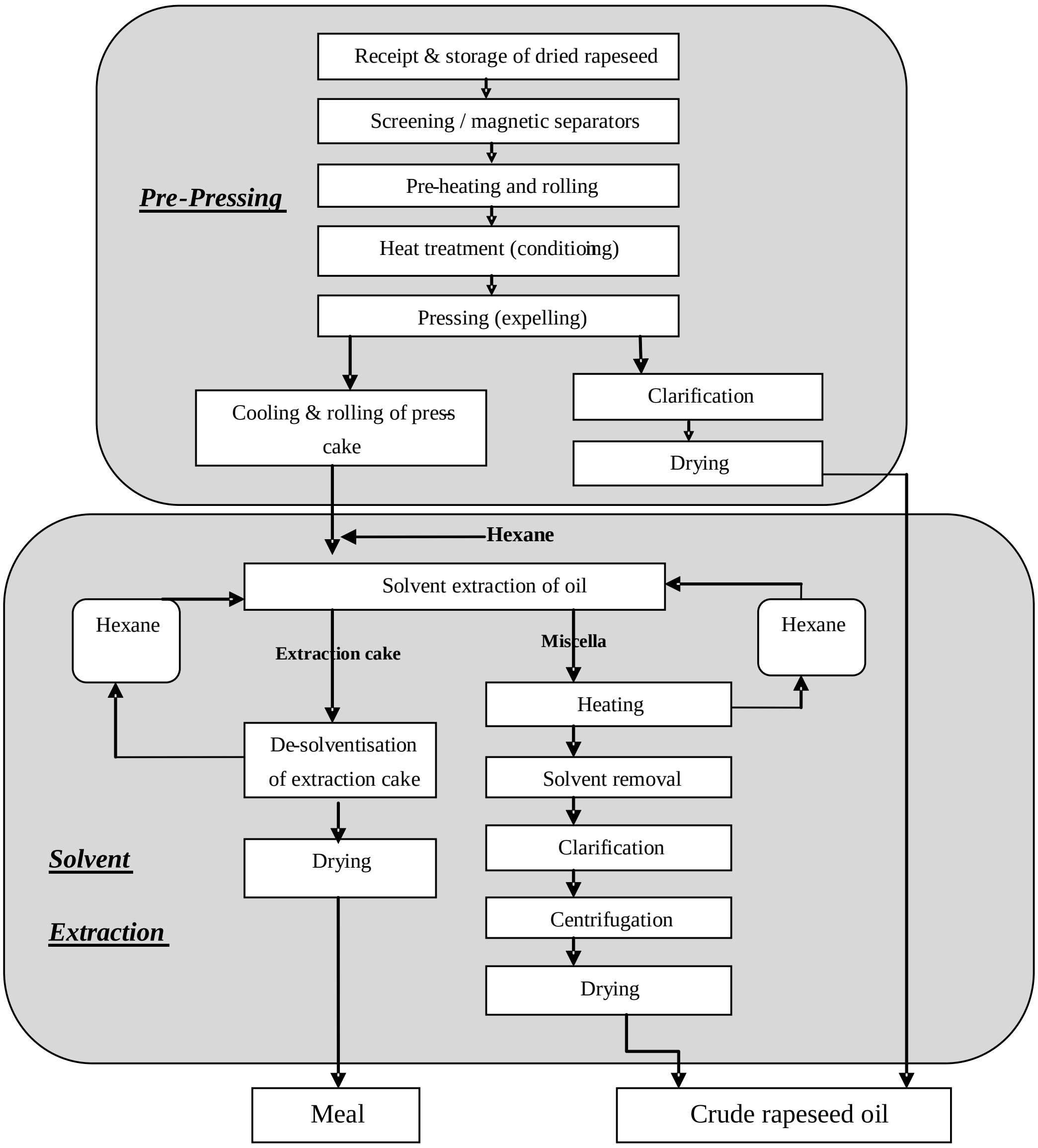

Figure 2.4-1: Main process stages for extraction of rapeseed oil

The principle of solvent extraction has remained largely unchanged since the practice was first used commercially in the early 1900's (Erickson and Wiedermann (1989) in Li et al., 2006). Continuous counter-current extraction is the most common process and Bockisch (1998) states that hexane is the solvent used most frequently for the extraction process, with minor solvents such as benzene, carbon disulphide and trichloroethylene also used. Hexane is most commonly used, being inexpensive, with good oil solubility at a relatively low temperature and non-corrosive to metal (Matthäus, 2007).

The output from solvent extraction, a mixture of oil and solvent called 'miscella', is separated by distillation into two components, crude oil and solvent, allowing the solvent to be recycled into the extraction process (FEDIOL, 2002). The meal must also have the solvent removed and this is performed through evaporation in a desolventizer-toaster. The combined recovery results in a solvent loss of approximately 0.2% to 0.3%. McManus et al.(2004) note that with the introduction of the UK Environmental Protection Act (1990) there is a legal requirement to maintain hexane use below 2 kg per tonne of processed seed.

The crude oil produced, contains a variety of undesirable compounds such as phospholipids, free fatty acids (FFA), pigments and volatile compounds, in addition to residual pesticides and fertilisers from agriculture. It must therefore be refined to improve oil quality for human consumption (Matthäus, 2007; Chumsantea, 2012).

Refining is a multistage process which generally takes place in the same facility as the extraction has been performed. It can use chemical or physical means, with the basic refining steps being: degumming, neutralising, bleaching and deodorising as shown in figure 2.4-2.

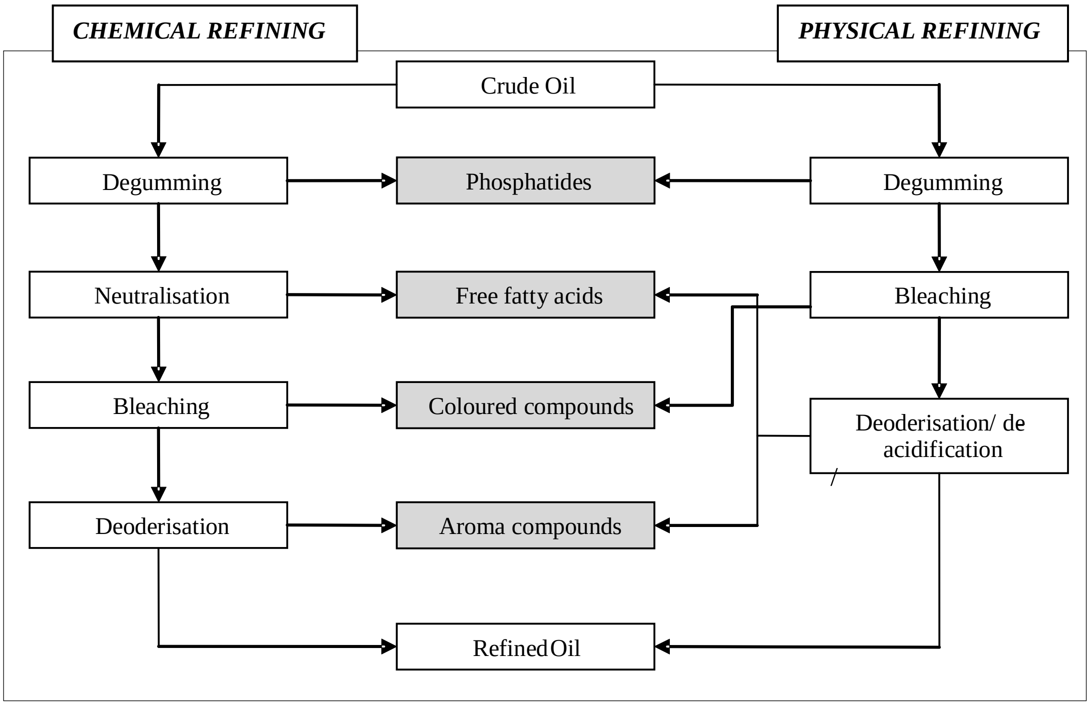

Figure 2.4-2. Basic steps for refining of edible oil. Source: Matthäus (2007)

2.5. **EDIBLE OIL EMULSIONS**

As outlined in section 1.3, emulsions are materials consisting of two immiscible liquids, with one dispersed in the other in the form of small spherical droplets. A large proportion of edible oil production is consumed as emulsion food products which include mayonnaise, butter, margarine, creams, ice-cream, sauces, desserts and cheese (McClements, 2005).

The two main types of emulsion are oil-in-water (O/W) emulsions or water-in-oil (W/0) emulsions. Within O/W emulsions such as mayonnaise, droplets of oil (lipophilic phase) are dispersed within an aqueous continuous phase (hydrophilic phase) as shown in figure 2.5-1 (Eisner, 2007).

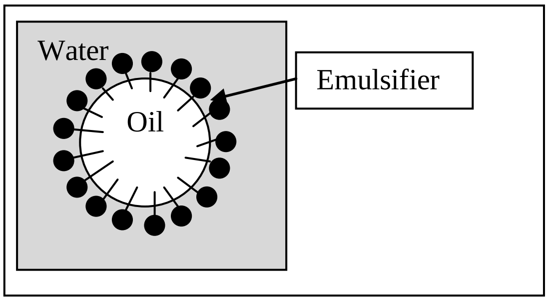

Figure 2.5-1: Diagram of O/W emulsion. Source: Eisner (2007)

There are several techniques for the manufacture of emulsions, with Eisner (2007) noting that the technique chosen depends on emulsion system and the standards expected of the emulsification process. All techniques have the same basic requirements however, with the first stage being generation of a course emulsion through agitation to create large droplets. This is followed by the application of mechanical energy to break up the oil droplets and introduction of a surfactant (emulsifying agent) to cause them to become re-encapsulated by the emulsifier.

Manufacture of edible oil emulsions requires the encapsulation of refined edible oil, using an emulsifying agent to generate the emulsion. For an O/W emulsion such as mayonnaise, egg yolk is used as the emulsifier with the minimum oil content specified as at least 70% by weight with 5% egg yolk (Dubruille, 1996 in Gunstone et al., 2007).

2.5.1. **Mayonnaise production**

Although potentially the simplest of all food emulsion products, mayonnaise is probably one of the most widely used condiments in the world today (Depree and Savage, 2001) and was ideal for use as the case study for investigation in this research. It is an O/W emulsion, with the type of oil used varying, dependant on brand and geographical location. Mayonnaise can be produced in either batch or continuous processes, with large scale mayonnaise production normally carried out using plant specifically designed for that purpose, which is most often semi-automated (www.edge.silverson.com, 2013).

Several equipment manufacturers include an outline of the steps involved in mayonnaise manufacture within their literature (www.edge.silverson.com, 2013; www.spx.com, 2013) together with Downing (1996) who outlines the basic steps for mayonnaise preparation. The first step is the preparation of the aqueous phase by dispersing the surfactant (egg yolk), in water, along with all other ingredients apart from the oil. For batch manufacture, this is typically performed in a mixing vessel, whereas for continuous manufacture metering pumps are used to feed each ingredient at the correct flow-rate through an in-line mixer. The aqueous phase is then mixed with an equal volume of oil to form a crude emulsion, before being fed to a colloid mill or homogeniser, during which time the remaining oil is incorporated as required, to form the required emulsion. The final product is then passed on to bottling and further packaging stages, prior to distribution.

A simplified flowchart of the process is shown in figure 2.5-2 in which the water required can be added either as a diluent for the vinegar, or as a brine (salt) solution.

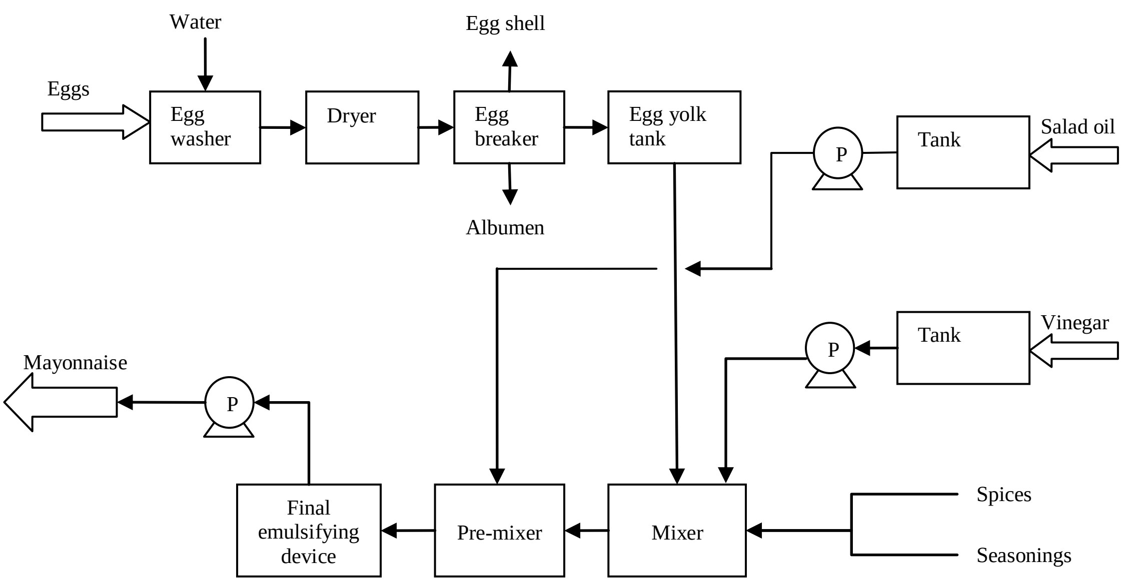

Figure 2.5-2: Schematic of typical conventional mayonnaise production flow. Source: Takashi Y, Hiroko M. (1999)

2.6. **OIL-BODIES**

In the introduction it was outlined that the overarching aim of the work presented by this thesis is to identify whether the new, novel process route for production of edible oil emulsions from oil-bodies has a better environmental profile than that of the existing technology route which involves extraction and refining of edible oil followed by re-encapsulation to form an emulsion. It is therefore important to understand a little of what oil-bodies are.

Organelles are the constituent parts of cells that perform a specific function, for example, nuclei store genetic information and mitochondria produce chemical energy. The organelles that store oils inside mature oilseeds are called oil-bodies, with Murphy (2001) noting that oil-bodies of various types are known by a plethora of different names in literature including lipid bodies, lipid droplets, lipid particles, lipid–protein particles, lipid globules, lipid inclusions, lipid vacuoles, lipoproteins, spherosomes, elaiospheres and oleosomes.

These oil storing organelles are found throughout most if not all plant cells, however they are most abundant in plant seeds (Tzen and Huang , 1992) where their primary function is to store neutral lipids during seed dormancy (Fisk et al., 2006) to provide a source of energy and structural fatty acids for the developing embryo (Christie, 2011). True oilseeds accumulate oil-bodies as one of their major storage reserves in amounts ranging from around 20% seed weight in soybean to 42% in rapeseed and as high as 76% in some of the larger seeded nuts (Murphy, 2001).

2.6.1. **Characteristics**

Oil-bodies (OBs) are around 0.5 – 3μm in diameter and consist of a central matrix of triacylglycerides (triglycerides or TAGs) surrounded by a single phospholipid (PL) layer (van der Schoot et al., 2011) and a layer of structural protein oleosin Huang (2011). Figure 2.6-1 gives an indication of oil-body size and structure, with a transmission electron micrograph (TEM) of a Brassica seed on the left hand side, showing the large and conspicuous storage protein bodies and small but numerous oil-bodies. The right hand of figure 2.6-1 shows a model of the structure of an OB, showing the three different structural components.

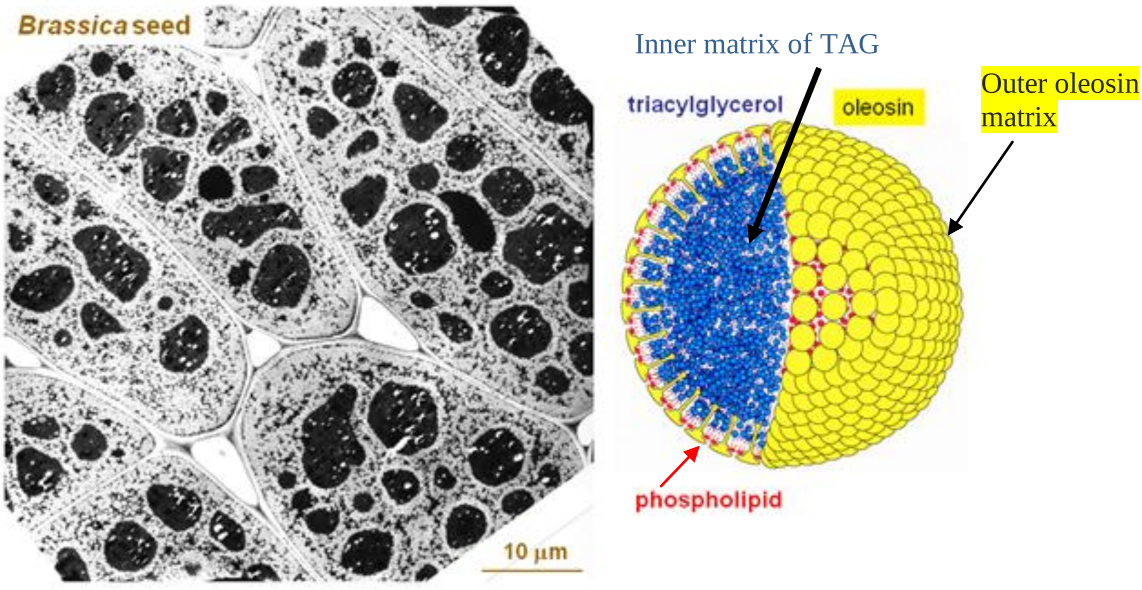

Figure 2.6-1: Oil-bodies in Brassica seed (left) and model of structure (right). Source: Huang (2011)

Triacylglycerols are the fatty acid triesters of glycerol for which Christie (2011) notes '*nearly all the commercially important fats and oils of animal and plant origin consist almost exclusively of the simple lipid class – triacylglycerols'* The resultant layered structure of the OB protects the oil within the organelle for optimum stability during dry storage and mobilisation during seed germination (Murphy, 2001).Tzen and Huang (1992) note that oil-bodies are remarkably stable either inside the cells or in isolated preparations, with Huang (1994) noting that they do not aggregate or coalesce, a property that as noted by Bhatla et al. (2010) makes them a suitable agent for emulsification.

2.6.2. **Past research**

Several papers have outlined techniques for the aqueous extraction of oil-bodies (OB) from a variety of oilseeds to generate a natural oil-in-water emulsion. During this process, which typically involves physical homogenisation or enzyme assisted digestion of the seed cell wall, the soluble cellular materials from the seed dissolve, allowing the release of oil into the bulk liquid phase. The oil can then be recovered from this phase by centrifugation resulting in a natural oil-in-water cream emulsion.

Physical OB extraction commonly consists of homogenisation using a blender (Tzen and Huang, 1992; Fisk et al., 2006; White et al., 2008; Nikifordis and Kiosseoglou, 2009), with the homogenisation medium depending on researcher and type of seed. Enzymatic extraction can be adopted as an extraction process alone or as a pre-treatment process followed by a physical extraction step (Campbell and Glatz, 2009).

Whether physical or enzymatic means are used, the OBs are isolated by first filtering the homogenate, then centrifuging the filtered material to liberate the oil-bodies in a cream. (Adams et al.,2012), which is a naturally occurring emulsion.

2.6.3. **Potential uses**

Several researchers have highlighted the potential for exploiting the natural oil-in-water emulsion generated as the product of the aqueous OB extraction as a more environmentally friendly or sustainable production route for the preparation of food products based on emulsions (White et al.,2008, Nikiforidis et al., 2012). Bhatla et al. (2010) cite the numerous potential uses including food and feed, pharmaceutical, personal care and industrial products. They go on to note that the use of oil-bodies extracted from plants such as sunflower as an emulsifying agent for food products may provide a healthier and potentially more economical alternative, since they are rich in polyunsaturated fatty acids and vitamin E. Examples of such products are cited as mayonnaises, ice creams, vinaigrettes, salad dressings, puddings, juices, icings, fish food, pet food and livestock feed.

2.6.4. Perceived benefits over conventional technologies

The creation of emulsions using aqueous extraction is perceived as having several potential advantages over the traditional milling and solvent extraction process (Campbell and Glatz, 2009; Nikiforidis et al., 2013). The aqueous extraction is anticipated to have far lower energy consumption than that required for the expelling, solvent extraction and refining stages, which could potentially lead to reductions in greenhouse gas (GHG) emissions in addition to reduced fossil fuel depletion.

In addition, the novel route removes the need for a toxic and inflammable organic solvent as part of the process, thereby reducing the health and safety risk of associated production equipment, in addition to potentially reducing the impacts on photochemical oxidant formation, which give rise to smog. The new process is also perceived to have fewer processing steps and this should reduce both energy intensity and raw material requirements.

The level to which these perceived benefits can truly be attributed to the new process proposed can however only be verified by systematic environmental assessment using LCA. This is the main driver of the work detailed by this thesis.

2.7. CONCLUDING SUMMARY

This chapter outlined key background information pertaining to the products and processes involved within this research.

An overview of edible oils was provided, with their characteristics, cultivation and processing discussed. Their use within food grade emulsions was outlined, with further detail provided on the production of mayonnaise, the emulsion product to be used as a case study comparator for the research. This led on to the introduction of oil-bodies, indicating their characteristics, a brief overview of work performed to date on their extraction and uses and an outline of their perceived benefits over conventional materials and processing.

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CHAPTER 3. LIFE CYCLE ASSESSMENT – INTRODUCTION AND APPLICABILITY

Following on from the overview of the key elements integral to the system being researched provided in the previous chapter; this chapter introduces Life Cycle Assessment (LCA), together with its single issue variant Carbon Footprinting (CFP), both of which will be used as the primary methodological tools to identify the environmental burdens of the systems outlined in this thesis.

As a tool designed to assess the environmental performance of a product or process by analysing the inputs and outputs throughout the entire life-cycle, the comprehensive nature of LCA is useful for preventing burden-shifting, whether from one phase of the life-cycle to another, from one region to another, or from one environmental problem to another. As such, there is general consensus on LCA's suitability as an effective tool for determining environmental performance (Finnveden et al. 2009).

This chapter will present a summary of the history and background of the development of LCA, together with an overview of the steps required. A summary of past usage will also be provided, to illustrate the relevance and applicability of LCA for fulfilling the aims of this research.

3.1. **HISTORY**

Many authors cite different founders for Life Cycle Assessment (LCA), with Astrup Jensen et al. (1997) noting that '*like all good ideas, LCA probably started in a number of different places, in a variety of different ways*'. McManus (2001) indicates that Boustead (1996) cites the origins as the UK during the energy crisis in the 1970's, whilst Baumann and Tillman (2004) note that many sources cite the Midwest Research Institute in the US as the birthplace of LCA (Guinee, 1995; Klopffer, 1997; Weidema, 1997). The earliest reported use of LCA type studies is recounted by U.S. EPA (2006) which states that one of the earliest publications of its kind was conducted by Harold Smith, project general manager for the Douglas Point Nuclear Generating Station, Canada. At the World Energy Conference in 1963, Smith reported his calculation of cumulative energy requirements for the production of chemical intermediates and products (ibid).

Later in 1960's several global modelling studies were conducted which led to further uptake of the early resource analysis techniques (the pre-cursors to LCA). In particular the publication of 'The National Academy of Science's Resources and Man' (1969), Meadows' book 'The Limits to Growth' (1972) and Club of Rome's document 'A Blueprint for Survival' (1972) resulted in predictions of the effects of the world's changing population and the expansion of industrial processes on demand for finite raw materials and energy resources.

Interest and usage grew steadily from the late 1960's and '70's with a rapid growth in interest during the 1990's when as noted in Finnveden et al. (2009) the first scientific publications emerged. At the time its results were often criticised (ibid) with U.S EPA (2006) noting that in 1991, concerns over the inappropriate use of LCA results for product marketing claims led to a statement on its use being issued by eleven State Attorney Generals in USA. The statement denounced the use of LCA results to promote products until a uniform methodology could be developed and a consensus reached on an appropriate manner for advertising such environmental comparison data in a non-deceptive way.

Responding to such concerns, the Society for Environmental Toxicology and Chemistry (SETAC) held the first international meetings for researchers and practitioners in 1990 and 1991 (Astrup Jensen and Postlethwaite, 2008) and continued to organise working groups and publish reports on various aspects of LCA methodology throughout the 1990s (McLaren, 2010). In parallel with this, the International Standards' Organisation (ISO) began work on a set of standards to harmonise the methodology resulting in the development of a set of four international standards (ISO 14040 to 14043) published between 1997 and 2000. These standards have subsequently been updated as shown in table 3.1-1.

Table 3.1-1: Overview of current and previous standards for LCA

Standard	Title	Status
ISO 14040:2006	Environmental management. Life cycle assessment. Principles and framework	CURRENT STANDARD Introduced in 2006 with ISO 14044, to jointly replace ISO 14040:1997, 14041:1998, 14042:2000 and 14043:2003
ISO 14041:1998	Environmental management. Life cycle assessment. Goal and scope definition and inventory analysis	Withdrawn in 2006, following publication of new versions of 14040 & 14044
ISO 14042:2000	Environmental management. Life cycle management. Life cycle impact assessment	Withdrawn in 2006, following publication of new versions of 14040 & 14044
ISO 14043:2000	Environmental management. Life cycle assessment. Life cycle interpretation	Withdrawn in 2006, following publication of new versions of 14040 & 14044
ISO 14044:2006	Environmental management. Life cycle assessment. Requirements and guidelines	CURRENT STANDARD As for ISO 14040:2006
ISO/TR 14047:2012	Environmental management - Life cycle impact assessment - illustrative examples on how to apply ISO 14044 to impact assessment situations.	CURRENT TECHNICAL REPORT. Replaced ISO/TR 14047:2003
ISO/TS 14048:2002	Environmental management. Life cycle assessment. Data documentation format	CURRENT TECHNICAL SPECIFICATION
ISO/TR 14049:2012	Environmental management. Life cycle assessment. Illustrative examples on how to apply ISO 14044 to goal and scope definition and inventory analysis	CURRENT TECHNICAL REPORT. Replaced ISO/TR 14049:2000

With increased interest in the application of LCA, the United Nations Environment Programme (UNEP) and SETAC launched their joint organization 'Life Cycle Initiative' in 2002 as an International Life Cycle Partnership '*to enable users around the world to put life cycle thinking into effective practice'* (www.lifecycleinitiative.org, 2013). The activities of the Life Cycle Initiative were divided into 5-year phases, with the first phase (2002-2007) focusing on activities aimed at moving the life cycle agenda forward by concentrating on three areas – life cycle management, life cycle inventory and life cycle impact assessment. The second phase which ran from 2007 to 2012 focused on the involvement of global stakeholders to reach a common understanding and agreement in areas such as life cycle assessment databases. The current phase, which started in 2012 and is due to run until 2016, aims to build on that platform and reach consensus in areas such as data, methods and product sustainability reporting (ibid).

As concern over climate change grew through the 1990's and 2000's, interest in the single-issue LCA variant, CFP flourished. CFP is an LCA that focuses exclusively on the greenhouse gas (GHG) results. Lillywhite (2010) states that '*carbon footprinting has a short but energetic history'* , however Finkbeiner (2009) clarifies this by explaining that whilst CFP is perceived as a recently developed tool, *'the concept of carbon footprinting has been in use for several decades but known differently as life cycle impact category indicator global warming potential'*. Having evolved from the establishment of the Intergovernmental Panel on Climate Change (IPCC), the world's leading international body for the assessment of climate change, the carbon footprint originally referred to only the emission of carbon dioxide, however it is now mostly applied to a normalised summation of all recognised GHG (Lillywhite, 2010).

Similar to the growth of LCA, it soon became apparent that guidelines and standards needed to be developed to ensure that practitioners could develop credible CFPs using standard methodologies and harmonised approaches for a plethora of choices such as boundary setting, inclusion and exclusion of emissions and for GHG reporting. Pandy et al. (2011) outline some of these guidelines and standards, including ISO14064 (2006) a three part standard that addresses the quantification and reporting of greenhouse gas emissions and the verification of this information and PAS 2050 (2008), which was the world's first standard for carbon footprinting of goods and services. The latter has since been superseded by PAS2050:2011. ISO have also been working on a standard for carbon footprinting, which has encountered numerous delays in its development, but has just been released as a Technical Specification; ISO/TS 14067:2013, entitled 'Greenhouse gases – Carbon footprint of Products – Requirements and guidelines for quantification and communication.'

3.2. LCA PHASES

LCA is defined by ISO 14040:2006 as the '*compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its lifecycle'*. The standard goes on to state that LCA models '*the life cycle of a product as its product system, which performs one or more defined functions'*. A product system is the collection of unit processes that perform one of more defined functions, which when put together model the product lifecycle. The standard methodology for LCA comprises four distinct phases as shown in figure 3.2-1, which will be described in the following sections.

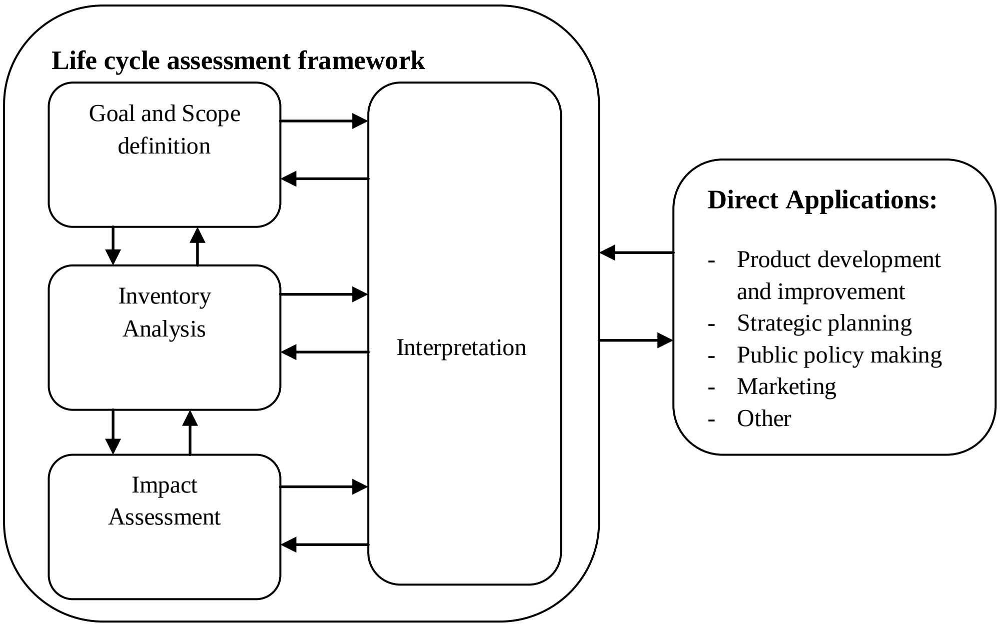

Figure 3.2-1: Stages of an LCA (ISO, 2006a)

3.2.1. **Goal and Scope definition**

The first stage of any LCA is the clear definition of goal and scope. ISO 14040:2006 states that the "*goal of an LCA study shall unambiguously state the intended application, the reasons for carrying out the study and the intended audience*". As such, an LCA practitioner must identify what is to be accomplished with the LCA, how the results will be used and who will be using them. The goal sets the framework for the study to be performed by describing the product system in terms of its boundaries, purpose and functional unit (FU), which is a key term and parameter for any LCA.

The FU forms the reference unit for assessment. It must be specific, measurable and clearly define the function of the product system to be assessed such that all the inputs and outputs can be identified. A useful way to do this is by creating a flow diagram that incorporates all the unit processes with their associated flows. Having specified the system in this way, the system boundaries must then be set, to classify which processes will be included and which excluded from the system. Where a study encompasses all life cycle stages it is termed a 'cradle to grave' study, whereas a 'cradle to gate' analysis excludes the unit processes downstream of manufacture.

Having defined the goal, the scope can then be developed to enable the goal to be achieved. ISO 14040:2006 states that "*The scope should be sufficiently well defined to ensure that the breadth, the depth and the details of the study are compatible and sufficient to address the stated goal*".

The scope will include descriptions of the following requirements and parameters:

- The product system to be studied, together with its function, its functional unit and system boundaries
- Allocation procedures
- Impact assessment methodologies and types of impacts
- Interpretation to be used, including value choices and optional elements
- Data requirements, data quality requirements and all assumptions used
- Limitations, both of the approach and the specific circumstances
- Type of critical review
- Format of reporting.

Whilst formulation of the goal and scope are always the first step in any LCA, this does not mean that they are then set in stone, as all LCA studies tend to be iterative in nature. As such, the parameters set at this early stage will almost certainly need adjustment throughout the process as information becomes available. As indicated in figure 3.2-2, it is important to re-visit the goal and scope throughout the assessment process to ensure consistency throughout.

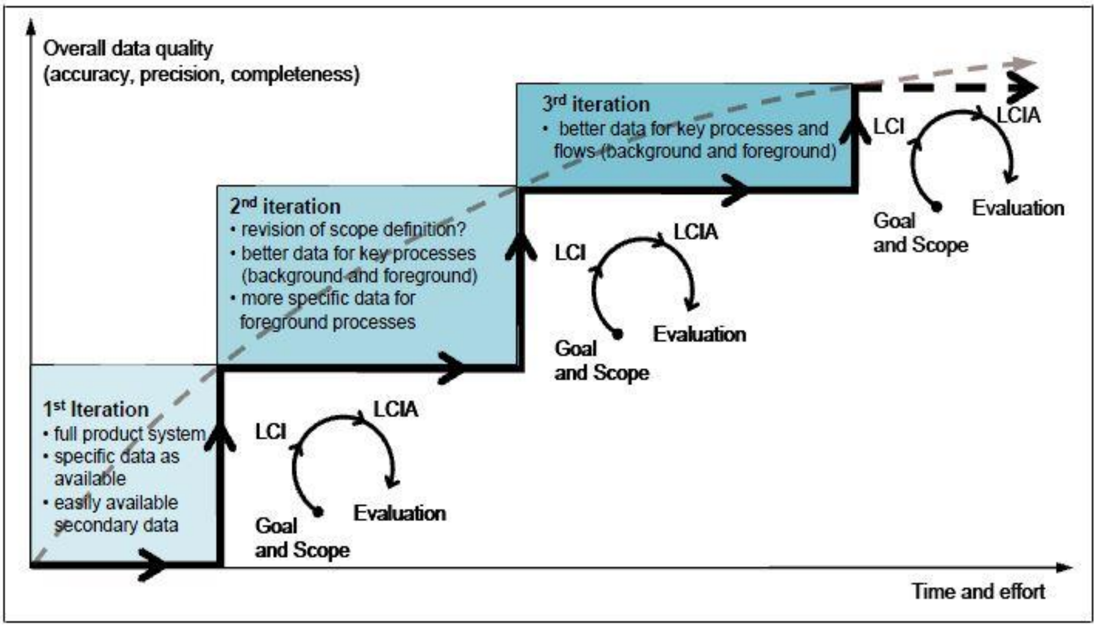

Figure 3.2-2: The iterative nature of LCA. Source: European Commission (2010)

3.2.2. **Inventory Analysis**

Having defined the goal and scope, the life cycle inventory (LCI) phase of the LCA is the phase where the actual data collection and modelling of the system (e.g. product) is done. This is performed in line with the goal definition and ensures that the requirements specified in the scope are met.

The inventory stage involves the collection of all the required data for the raw materials and energy inputs to the product system, together with the wastes and effluents generated at each stage; a process which as noted by Tillman (2010) is often the most time consuming activity when conducting an LCA study. Clift et al. (1998) note that it is often useful to divide the system into foreground and background systems, for which the foreground system comprises the activities or processes of direct interest that are delivering the functional unit, while the background system supports the foreground activities by supplying the necessary energy and materials (Baumann and Tillman, 2004; Azapagic et al., 2007). This distinction affects the type of data required, with specific process data required for the foreground system, while the background is normally represented by data for a mix or a set of mixes of different technologies or processes (Azapagic et al., 2007).

The volume of data collected and analysed is large and extensive, as is the inventory table developed, which represents all the inputs and outputs to the system. As outlined for the goal and scope determination, this stage is also most often iterative, with calculations frequently performed several times as the model progresses from less detailed to refined, in line with the specified data requirements. As such, for all but the most simple and streamlined LCAs, the data manipulation and analysis is performed by computer modelling, most often using a proprietary software package, designed specifically for LCA.

3.2.3. **Impact Assessment**

Extraction of any meaningful results from the inventory table generated by the LCI stage is difficult if not impossible without the third step in LCA, namely life cycle impact assessment (LCIA). The impact assessment stage links the inventory data with environmental impacts and attributes values to the potential magnitude of those impacts. An impact is the environmental change that results directly from the emission of a substance by the product system, e.g. the primary impact of the release of SO_2 into the air is acidification. ISO/TR 14047:2003 states that the purpose of LCIA is '*to assess a product system's life cycle inventory analysis (LCI) results to better understand its environmental significance*', this involves the sorting of inventory data into the relevant impact categories as specified by the scope.

The LCIA contains an unavoidable degree of subjectivity in the linkages and assessment of magnitude that cannot be eliminated. ISO 14040:2006 therefore differentiates between certain of the LCIA stages, stipulating that some are mandatory, whilst others are optional, as illustrated in figure 3.2-3. Those stages that are specified as optional have a higher degree of subjectivity and uncertainty and are considered the most contentious.

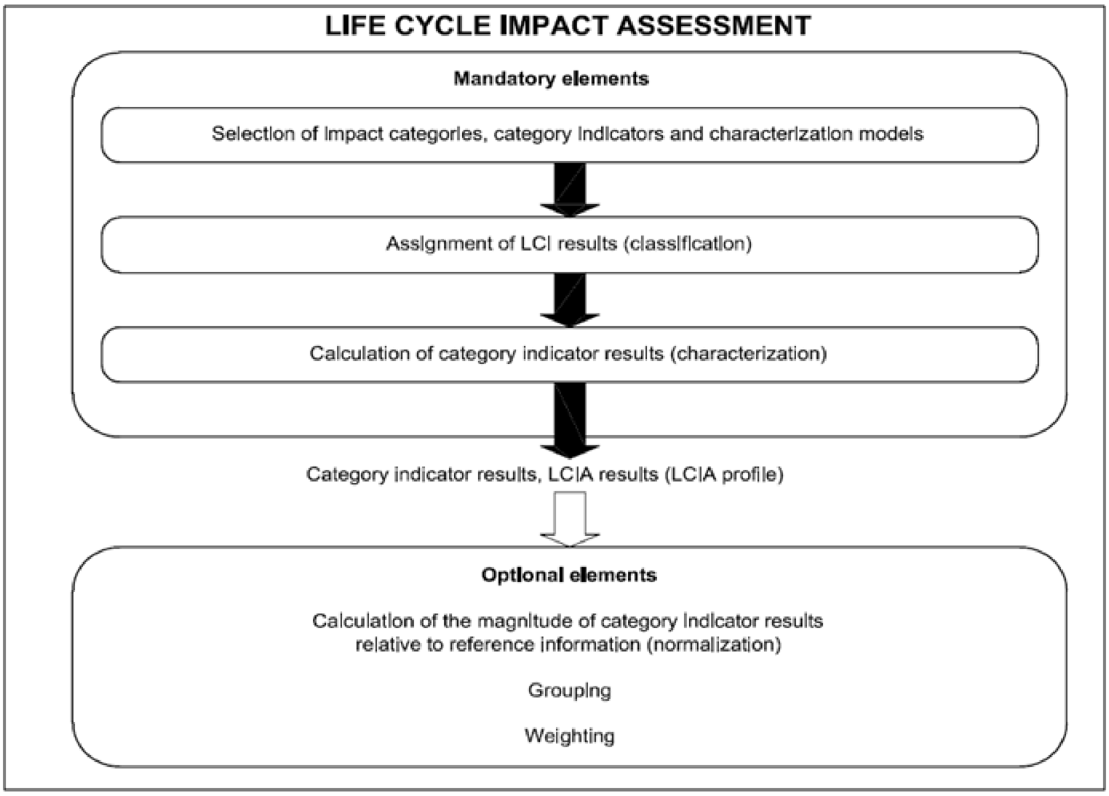

Figure 3.2-3: Optional and mandatory elements of LCIA (ISO 14040:2006)

The first stage of LCIA, category definition, entails the choice of categories to be used for linking the data. There are two main groups of choice for category indicators, termed Midpoints and Endpoints.

Midpoints place the indicators relatively close to the source of emission. Radiative forcing and PM_{10} concentration are both examples of these. Midpoints have the advantage of relying primarily on scientific information and well-proven facts and as such, the amount of subjectivity and uncertainty involved is limited.

Endpoints place indicators relatively close to the endpoints of concern. These have the advantage of presenting information in an appealing and understandable way: human health is easier to interpret and communicate than ozone layer depletion or radiative forcing.

It is generally accepted (Baumann and Tillman, 2004) that the general endpoint categories of impact assessment for an LCA include ‘Resource use’, ‘Human Health’ and ‘Ecological consequences’. Each of these three general categories, which are often termed ‘damage categories’ or ‘areas of protection’ can be further subdivided into several specific impact categories such as ‘Global warming’, ‘Acidification’, ‘Human toxicity’ etc.

Impact categories are defined based on the requirements of the goal and scope of the study e.g. if it is defined in the goal and scope that the study is to be a carbon footprint assessment and thereby assess the emission of GHG's from the product system, the impact categories will reflect that and focus exclusively on GHG's.

Classification then links the inventory data with the chosen impact categories by sorting them according to the effect they have on the environment. This is followed by the last of the mandatory steps, characterisation, in which the potential magnitude of the emissions, which were grouped together during the classification step are identified by applying a characterisation factor.

These characterisation factors are dependent on the LCIA method chosen and for those methods incorporating midpoint and endpoint analysis, different characterisation factors will be applied to each step. Goedkoop et al. (2013) give the equation for applying the characterisation factors as follows:

$$I_m = \Sigma_i Q_{mi}\ m_i \qquad \textbf{Eq 3.2-1}$$

Where m_i is the magnitude of intervention i (e.g. the mass of CO_2 released to air), Q_{mi} the characterisation factor that connects intervention i with midpoint impact category *m*, and I_m the indicator result for midpoint impact.

In other words for each impact category, every relevant categorised inventory value e.g. amount of CO_2 or methane (within the climate change category) is multiplied by its respective characterisation factor, and these are summed together to produce the midpoint or endpoint indicator for that category. Figure 3.2-4 provides a useful overview of this process showing the interaction between midpoints and endpoints, using greenhouse gases and their climate change midpoints and endpoints as an example.

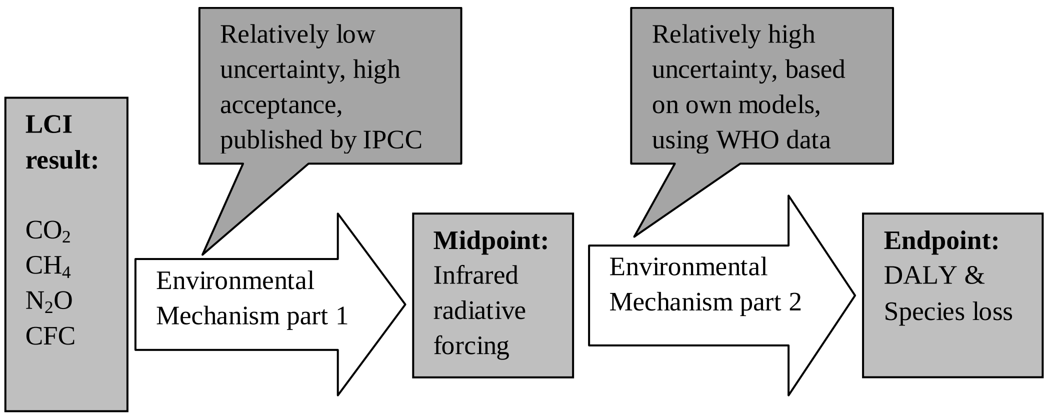

Figure 3.2-4: Example of midpoint-endpoint model for climate change, linking to human health and ecosystem damage. Source (Goedkoop et al., 2013)

As noted by Van Hoof et al. (2013) there have been many efforts over the years to simplify and improve the way that LCA results can be used for decision making. Whilst perusal of the raw characterised data can be useful for those who are able to interpret the significance of the results in their kg CO_2 equivalents or kg 1,4 DB (1,4-dichlorobenzene) equivalents form, evaluating such results can be problematic. The first of the optional steps, normalisation is designed to aid this interpretation, by identifying those impacts that differ greatly from the norm, in addition to facilitating the comparison of different impact categories that would not otherwise be equivalent due to different units..

It achieves this by dividing the impact category results from characterisation by a “reference” or “standard” value, e.g. the effects caused by the average European during a year. This emphasises the impact categories which have the greatest environmental effect, in comparison to the ‘norm’.

The most controversial step in LCIA is weighting, which if performed, assigns weights or relative values to the different impact categories based on their perceived importance or relevance (U.S. EPA, 2006). The weighting factors can be determined either using a distance to target approach, a panel of experts, or using a monetarisation approach. In the distance to target method, each impact category has a set target and the distance from that target defines the weighting (if the difference is high, the weight is high). ISO 14044:2006 stipulates that weighting, *‘shall not be used in LCA studies intended to be used in comparative assertions intended to be disclosed to the public’*.

3.2.4. **Interpretation**

The final stage of an LCA, interpretation, attempts to evaluate and report the findings of the LCIA or LCI or both based on the stated goal and scope of the study, to reach conclusions and recommendations. U.S. EPA (2006) identifies the key stages to interpretation as:

1) Identification of the prominent issues
2) Evaluation of the completeness, sensitivity and consistency of the data
3) Drawing conclusions and recommendations.

As noted by Adams (2011), these three stages help to provide a constructive, systematic approach to interpreting the life-cycle of the product system, in addition to establishing the confidence in and reliability of the results.

3.3. **ATTRIBUTIONAL AND CONSEQUENTIAL LCA**

Despite the standard LCA methodology outlined in the ISO 14040 series of standards, there is still considerable debate concerning several methodological choices. Reap et al. (2008)

present a survey of '*unresolved problems in life cycle assessment'* in their two part article (Reap et al., 2008 a and b) and Finnveden et al. (2009) provide a comprehensive outline of the recent developments focussing on '*areas with prominent methodological development during the last years.. [*together with*].. some of the emerging issues'*.

Thomassen et al., (2008) explain that several researchers have outlined two distinct approaches for LCA (Heijungs, 1997; Frischknecht, 1998; Ekvall, 1999; Tillman 2000; Weidema 2003). The two approaches are generally described as 'attributional LCA' and 'consequential LCA', terminology that as noted by Ekvall and Weidema (2004) was first adopted at a workshop on LCI electricity data in Cincinnati in 2001 (Curran et al., 2001 in Ekvall and Weidema, 2004). The terminology itself is not comprehensively accepted however, with other practitioners referring to the different LCA's as 'prospective' and 'retrospective' (Ekvall et al., 2005) or 'accounting' and 'change-oriented' (Baumann and Tillman, 2004).

Thomassen et al. (2008) summarise the description by Rebitzer et al., (2004) of the approaches, stating that attributional LCA ' *describes the pollution and resource flows within a chosen system attributed to the delivery of a specified amount of the functional unit'* , whereas consequential LCA ' *estimates how pollution and resource flows within a system change in response to a change in output of the functional unit'* (ibid).

Since the two approaches were developed in the process of resolving methodological debates surrounding treatment of co-products (ibid), the manner in which they account for co-products differs. Thomassen et al. (2008) explain that with attributional LCA, the use of system expansion to handle co-products is optional and in most cases allocation is used in preference. This is discussed fully in section 4.1.2 where the ISO recommended approach for determining treatment of co-products is outlined. This differs from the consequential approach, where use of system expansion is the only manner to deal with co-products as it reflects the consequences of a change in production.

In addition to this, Schmidt (2010a) outlines another methodological difference, stating that consequential modelling includes the processes that are actually affected as a consequence of the decision the LCA is aiming to support (sometimes called 'marginal supply'), whereas attributional LCAs include the market average supply.

Finnveden et al. (2009) outline some of the current debate, stating that Lundie et al. (2007) argue that consequential LCA should be used for decision making, apart from when the difference between consequential and attributional LCA results is small. They go on to outline that attributional LCA should be used where no decision is at hand, a statement supported by Whittaker et al. (2011) who argue that consequential LCA is better suited to policy analysis.

Ekvall et al. (2005) state that attributional and consequential LCA can both be used for decision making and learning purposes in addition to the modelling of future systems. Furthermore, Hospido et al. (2010) state that based on work by Sandén and Karlström (2007), 'prospective attributional' LCA is assumed to be the most suitable approach to evaluate novel systems. In light of the increased complexity introduced by a consequential LCA, and the requirement to complete multiple LCAs, some of which concern novel systems within this project, the advice from both articles was heeded and attributional was chosen as the methodology for each study.

3.4. USE OF LCA RELEVANT TO THIS THESIS

For the systems investigated in this thesis, a review of available LCA and CFP studies pertaining to each relevant product system was performed, using the bibliographic databases available to The University of Bath. These included Compendex, Web of Knowledge and Scopus, together with standard internet searching where applicable.

The acquisition of such information enabled verification of LCA as a suitable tool for the required analysis by showing that such methodologies had been used previously for the identification of environmental impacts within product systems for each of the relevant areas. In addition, the review of articles detailing previous LCA and CFP studies enabled an understanding of the scope, application, sensitivities and limitations of past work to be obtained.

Information from the articles concerned was also of benefit for identifying the process flows for the systems involved, accessing and verifying LCI data, building understanding of the complexities in modelling novel processes and where possible, validating results.

3.4.1. Usage within food products

Life Cycle Assessment (LCA) has been used widely as a tool to quantify the full range of environmental impacts of systems across the supply chain including food products. Anderson and Ohlsson (1998), Foster et al. (2006), Schau and Fet (2008) and Williams et al. (2010) all provide information on the multitude and variety of LCA studies performed in this sector. Roy et al. (2009) contains a useful overview of a number of LCA studies involving food systems and note that whilst the majority of LCA studies up to that point (2009) concerned industrial processes, a number have been applied to food products, although predominantly the industrial food products, a view supported by Milà i Canals et al. (2011) who also note that LCA coverage in terms of product groups is mainly concentrated on a few distinct subsectors '*e.g., meat, milk, a few cereals, some vegetable oils'*. Notarnicola et al. (2012) emphasise how important it is that '*we do more integrated LCA studies with regard to our entire food* *production and consumption system'*, a sentiment supported by McLaren (2010), who state that '*it is clear that life-cycle thinking is critical in the evaluation of alternative options for more sustainable food systems'*.

As noted by McLaren (2010) and Peacock et al. (2011) a widening range of initiatives to provide information about the environmental performance of food and drink products has recently been observed. In recent years, an increased focus on greenhouse gas (GHG) accounting over the entire supply chain has been fostered by such initiatives as the UK 'Carbon Label' and Sweden's 'Klimatmärkning', borne out of a desire to fulfil GHG reduction commitments. This strong focus on carbon reduction has resulted in the popularity of the single-issue LCA variant carbon footprinting (CFP) soaring.

As such, it is clear that both CFP and full LCA are established tools for the identification of environmental impacts of food systems and the results of this analysis will provide useful data to researchers in the field, in addition to the wider food and drink community.

3.4.2. **LCA of edible oils and their seeds**

LCA has been used widely for the assessment of edible oil systems and there are a considerable number of publications concerning the environmental impacts of products containing both rape and sunflowerseed oils, including margarines (Nilsson et al., 2010) and biofuels (Halleux et al.; 2008, Cocco, 2011; Iriarte et al., 2010; Thamsiriroj and Murphy, 2010). Of the studies identified, only four outlined the environmental performance of either seed oil as a product (McManus et al., 2004; Narayanaswamy et al., 2004; Schmidt, 2010; Roïz and Paquot, 2013) however and where such results have been shared, these have involved using different variables and methodological choices within the analysis.

Milazzo et al. (2013) provide a useful review of LCA studies pertaining to rapeseed derived bio-diesels, within which the general characteristics of 27 studies are outlined. Each of these studies, together with several additional publications concerning LCAs of rape and sunflowerseed oils were reviewed to assess relevance of data, and usefulness for the research presented here. The findings from those studies identified as relevant are summarised in table 3.4-1, which was used to assist in the choice of certain key methodological parameters in chapter 4.

Two studies were found specifically assessing the edible oils for food use (Narayanaswamy et al., 2004; Schmidt, 2010b) and whilst LCA is an accepted analysis tool for foodstuffs, no published data could be found concerning the case-study emulsion mayonnaise. Two very useful articles were found describing LCA studies on spreads and margarine (Schonfeld and Dumelin, 2005 and Nilsson et al.,2010) which is another edible oil emulsion. The data and findings of these are relevant for this project as the LCA results could in part be used to corroborate results from the LCA modelling.

Table 3.4-1: Summary of LCA studies performed on rape and sunflowerseed oils

Reference	Application	Oil	Location	Allocation	Software	LCIA	Comments
Schonfield and Dumelin (2005)	Food: margarine	R&S	Germany/ Netherlands	Economic	--	--	
Narayanaswamy et al. (2004)	Food: canola oil	R	Australia	Economic	SimaPro	CML 2002	
Nilsson et al. (2010)	Food: margarine	R&S	Germany/ Netherlands	Economic / mass	SimaPro	CML 2001	Sensitivity analysis for allocation
McManus et al. (2004)	Lubricants: Hydraulic fluid	R	UK	Mass	SimaPro	EI- 95	
Bernesson et al. (2004)	Energy: biodiesel	R	Sweden	Economic/ mass/ system expansion	--	--	Sensitivity analysis for allocation
Halleux et al. (2008)	Energy: biodiesel	R	--	Credits	SimaPro	EI-99	
Reijnders and Huijbregts (2008)	Energy: Biodiesel	R	Europe	Economic	--	--	
Stephenson et al. (2008)	Energy: Biodiesel	R	UK	Economic	Gabi	EDIP 2003	
Chiaramonti and Recchia (2010)	Energy: biofuel	S	Italy	Credits	GEMIS	IPCC 2007	
Iriarte et al. (2010)	Energy: Biodiesel	R&S*	Chile	n/a	Gabi	CML2001	Seed only
Schmidt (2010b)	Food/fuel: Edible Oil	R	Denmark	System expansion	SimaPro	EDIP 97	
Stephenson et al. (2010)	Energy: Biodiesel	R&S	South Africa	Economic	Gabi	EDIP 2003	
Thamsiriroj and & Murphy (2010)	Energy: Biodiesel	R	Ireland	Energy content	--	--	
Arvidsson et al. (2011)	Energy: Biodiesel	R	Sweden	System expansion	--	--	
Cocco et al. (2011)	Energy: Biodiesel	R	Southern Italy	System expansion	Boustead model	--	
Reinhard et al. (2011)	Energy: Biodiesel	R	Switzerland	System expansion	--	CML, UBP06	
Sanz Requena et al. (2011)	Energy: Biodiesel	R &S		Not specified	--	EI-99	
Gasol et al. (2012)	Energy: Biodiesel	R	Spain	System expansion	SimaPro	CML	
Nanaki and Kroneos (2012)	Energy: Biodiesel	R	Greece	Not specified	SimaPro	EI-99	
Spinelli et al. (2012)	Energy: Biodiesel	S	Italy	Mass	SimaPro	EI-99	
Spugnoli et al. (2012)	Energy: Biodiesel	S	Italy	Energy Content	Biograce	Biograce / IPCC	
González-García et al. (2013)	Energy: Biodiesel	R	Spain	Economic	SimaPro	CML	
Iriarte et al. (2013)	Energy: Biodiesel	S	Chile	Energy content	Gabi	CML	
Roïz and Paquot (2013)	Lubricants: hydraulic fluid	R	Belgium	Mass **energy content	--	Composite **CML / ReCiPe & EDIP2003	Sensitivity analysis for allocation & LCIA using '**'

R= Rapeseed; S = Sunflowerseed

3.4.3. **Use within novel technologies**

As stated in Hetherington et al. (2013), whilst a myriad methodological challenges are debated by the LCA community (Ekvall and Weidema 2004; Roy et al. 2009), there is a general consensus on LCA's suitability as an effective tool for determining environmental performance (Finnveden et al. 2009) and it is used widely as a decision-making tool in process selection, design, and optimization (Azapagic, 1999; Del Borghi et al. 2007). Koller et al. (2000) and Tufvesson et al. (2013) note that full-scale LCA is often thought of as too difficult or time consuming to pursue at the research or development stage of a new product or process.

There are certainly a number of methodological and practical difficulties that arise from using LCA at this stage and Hospido et al. (2010) provide an extremely useful account of the methodological issues affecting LCA of novel food products. They outline some of the practical difficulties of developing an LCA for a process that is still in development, including the type of LCA to use, choice of functional unit, identifying system boundaries, issues with data gathering and development of scenarios.

Kunnari et al. (2009) discuss options for methodological changes, based on the work of Nielsen and Wenzel (2002) who advocate the use of a stepwise LCA procedure in parallel with the development process. Use of LCA in this way often entails the assessment of lab and/or pilot-scale processes to generate environmental load data, which can then be used to optimise the developing process (ibid). This is the approach that will be used for the research detailed by this thesis, where the data will also be compared with existing industrial processes, to demonstrate or identify any environmental advantages that the 'novel' process affords over the existing activities.

3.4.4. **Carbon footprinting as an environmental indicator**

Williams et al. (2012) note that CFP is one of the foremost methods available for helping tackle the threat of climate change through quantifying anthropogenic GHG impact. A search of the bibliographic database 'SCOPUS' on the term 'Carbon Footprint' clearly demonstrated the growth in its usage with an exponential rise in publications on the topic, from 15 in 2000, to 295 in 2008 when the first standard PAS 2050:2008 was published (since revised in 2011), to 1061 articles in 2012. However as noted by Weidema et al.(2008) the rise in popularity of CFP was not driven by research, but from promotion by nongovernmental organizations (NGOs), companies, and various private initiatives.

Whilst the simplification of environmental assessment afforded by CFP could have the potential for burden shifting both within life cycle stages and impact categories, limited literature exists comparing the results of CFPs and full LCAs to determine the suitability as an environmental indicator and for targeting process improvements. Rugani et al. (2013) analyse the use of CFP as an environmental indicator within the wine industry, however they do not provide comparisons between the results of the CFPs and wider LCAs.

Laurent et al. (2012) performed a statistical correlation of CFP results with 13 other impact scores for 4000 life cycle inventories (LCIs) derived from the Ecoinvent database v2.2. These encompassed products, technologies, infrastructure and services, with agricultural LCIs comprising around 2.5% of the correlated datasets. Rapeseed oil was included in the analysis, however sunflowerseed oil was not and neither were any variants of mayonnaise or any other food product.

Within their published results, they concluded that '*A genuine correlation between carbon footprint and all the other environmental impact indicators can be observed if and only if all impacts from the product life cycle predominantly stem from one or few key processes that covary.'* (Laurent et al. 2012). In addition, they found that the impact categories that correlated the least with CFP results were toxicity impacts, together with depletion of resources and land use.

The review of both CFP and full LCA results for the systems researched within this thesis will therefore clarify the suitability of CFP as an indicator of environmental performance that can be used to target process improvements for the systems studied.

3.5. **CONCLUDING SUMMARY**

Following on from the outline of the systems to be investigated presented in chapter 2, this chapter introduced the key methodological tool used for analysis, LCA, together with its single issue variant, CFP. Following a brief outline of the history of LCA development, an overview of the standard methodology was presented, together with an insight into the differences between attributional and consequential LCAs, which is an area of considerable controversy and debate by the LCA community.

Finally, a summary was provided of the range of published literature concerning LCA studies in the areas of edible oils, foodstuffs and novel products, together with a review of available literature comparing CFP studies with full LCAs.

CHAPTER 4. METHODOLOGY

Using some of the information from the literature already presented, the key methodological considerations and procedures used to fulfil the research objectives will be outlined in this chapter. The parameters and decisions involved when specifying the goal and scope of the Life Cycle Assessments (LCAs) required to fulfil the stated research aims will be set out and approaches for data collection and modelling discussed. The detailed specification of the required elements of each goal and scope will then follow in the relevant chapters.

4.1. DEVELOPMENT OF LCAS FOR THIS THESIS

The primary aim of the work presented by this thesis was to determine whether the novel processing route for production of food grade edible oil emulsions via aqueous extraction of oil-bodies (OB) had a better environmental profile than that of the existing technology route. This required a comparison between the naturally occurring oil-in-water (o/w) emulsion and a comparable material generated via conventional technology.

Since the OB material was neither functionally equivalent to edible oil nor a conventional emulsion product, its performance needed to be determined within such a product. This could then be compared against the performance of that same product manufactured via the conventional route. As one of the simplest and yet most popular emulsion products, mayonnaise was chosen as the case-study emulsion.

4.1.1. Goal and scope definition

To perform the comparative analysis, separate LCA models were constructed for evaluation of the following product systems:

i. Production of refined rape and sunflowerseed oils
ii. Production of mayonnaise produced with rape and sunflowerseed oils
iii. Production of food-grade aqueously extracted oil-bodies
iv. Production of mayonnaise-like oil-body emulsion

The functional units and boundaries for each of these will be detailed in the relevant chapters; however the methodological considerations and choice of common parameters will be discussed here.

4.1.2. Allocation rationale

Both rape and sunflowerseed oil processing systems involve the production of not only the product of interest, but also a co-product during both the extraction and refining stages. In both cases the co-product, which is meal in the extraction and acid-oil in the refining stage can be sold as protein meal for animal feed. In addition, within the oil-body (OB) extraction process, residues from extraction are expected to also be utilised as animal feed.

As briefly discussed in section 3.3, product systems such as this require the issue of allocation to be considered, to determine the proportion of the environmental impacts that will be attributed to the production of each product. Allocation is defined as *'partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems*' (ISO 14044:2006). ISO 14040:2006 states that allocation should be avoided where possible, in favour of system expansion, however as highlighted in the U.S EPA (2006), expansion of systems is not possible in all cases and it can be argued that choice of allocation method should be based on what type of LCA is being performed (Baumann and Tillman, 2004).

ISO 14044:2006 advocates a stepwise procedure for determining how to handle co-products within a system, as follows:

Step 1: Wherever possible, allocation should be avoided by

1. dividing the unit process to be allocated into two or more sub-processes and collecting the input and output data related to these sub-processes, or
2. expanding the product system to include the additional functions related to the co-products, taking into account the requirements of 4.2.3.3 (*of ISO 14044:2006*).

Step 2: Where allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products or functions in a way that reflects the underlying physical relationships between them; i.e. they should reflect the way in which the inputs and outputs are changed by quantitative changes in the products or functions delivered by the system.

Step 3: Where physical relationship alone cannot be established or used as the basis for allocation, the inputs should be allocated between the products and functions in a way that reflects other relationships between them. For example, input and output data might be allocated between co-products in proportion to the economic value of the products.

Guinée et al., 2004 note that although preferred, system expansion is not always practical, a view confirmed by Milazzo et al., (2013) who state that allocation approaches are less data - intensive and that *'in analysing farm-based processes economic, energy and weight allocations are the norm'*. From studying the published oilseed LCAs summarised in table 3.4-1, it was evident that for most seed oils, the favoured allocation method was economic.

The basis for this is that oil crops are harvested for their oil, without which, they would not be financially viable to grow (the exception being soy bean oil – which is primarily grown for animal feed from the meal).

The mass of oil produced is constrained by the oil content of the seed which is around 40% for rapeseed and between 35% - 45% for sunflowerseed (www.fediol.eu, 2013 (d) and (e)). As such, allocation by mass would always lead the vast majority of impacts to be assigned to the secondary product, with the primary product (the oil) being assigned only a small impact. It was therefore decided to use economic value as the primary allocation methodology.

Milazzo et al. (2013) note that '*as stated in the ISO 14040-44 series, whenever more than one allocation method can be applied, a sensitivity analysis is required'*. Since the choice of allocation approach can have a profound effect on the results generated (Curran, 2007, Halleux et al., 2008, Morais et al., 2010), the impact of choosing economic value as the primary allocation method was investigated fully. by performing a sensitivity analysis on the impact of using both mass allocation and system expansion in accordance with the requirements of ISO 14044:2006 *'to illustrate the consequences of the departure from the selected approach.*'.This will be reported in chapter 5, for the seed oil LCAs.

4.1.3. **Allocation methodology: Economic**

For calculation of the required parameters for economic allocation, the prices for rape and sunflowerseed oils were obtained from data available via the International Monetary Fund (IMF) website, which publishes historical data on market prices for a range of commodities, including rapeseed and sunflowerseed oils. Figure 4.1-1 shows the fluctuation in market prices for both oils since 1980, from which it is evident that there has been a step change in seed oil prices in the last 10 years. In order to utilise the most representative data it was decided to use an average of the last five years prices as the allocation price for the oil.

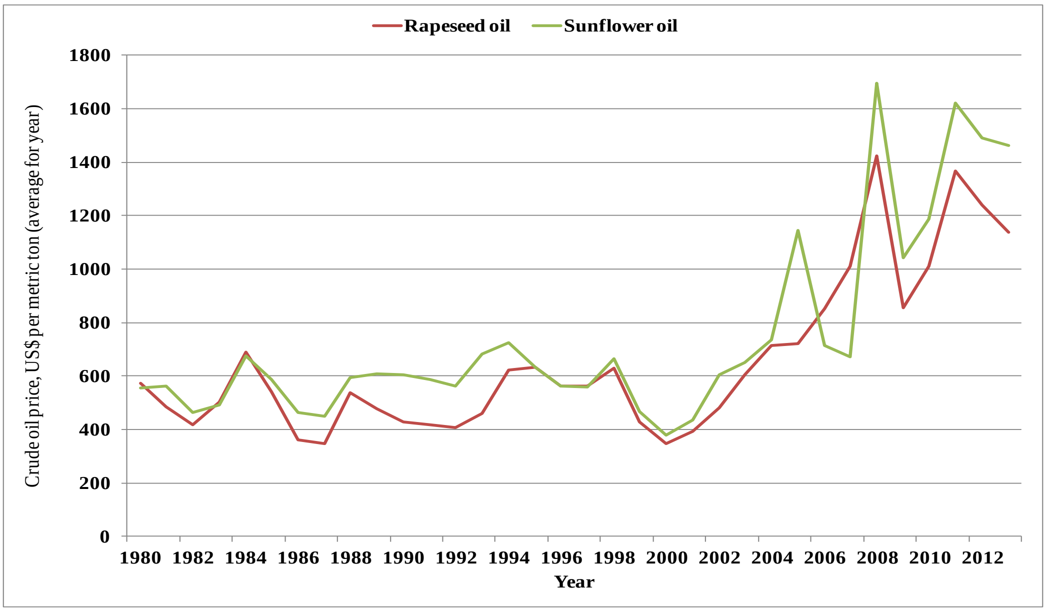

Figure 4.1-1: Crude rape and sunflowerseed oil prices since 1980 (www.imf.org, 2013)

Whilst the IMF data included prices for soybean meal, there were no prices indicated for rape or sunflowerseed oil meals generated as part of the seed extraction process. These were accessed instead from the 'DairyCo' division of the Agriculture and Horticulture Development Board in the UK. Details of the prices used are shown in table 4.1-1.

Table 4.1-1: Prices utilised for seed oils and meals as part of allocation process

	Unit	2008	2009	2010	2011	2012	Average
Rapeseed oil	$/ tonne	1423.71	856.18	1011.75	1366.63	1239.08	1151.53
Sunflowerseed oil	$/ tonne	1693.65	1041.6	1186.0	1621.8	1489.5	1284.27
Seed meals	$/ tonne	262.39	241.45	289.46	270.93	354.62	283.77

The allocation factors were then calculated by determining the value of the desired product (oil) compared with the value of the combined product stream (oil and meal). For the extraction process, 2500 kg of seed is required to produce 1000 kg of oil and 1500 kg meal, thus the allocation factor for the 1000 kg of oil was calculated according to equation 4.1-1, yielding allocation values as shown in table 4.1-2.

$$\frac{(1000 \times 1151.53)}{(1000 \times 1151.53) + (1500 \times 283.77)} = 0.7301 \qquad \textbf{Eq: 4.1-1}$$

Table 4.1-2: Economic allocation values used within rape and sunflowerseed oil LCAs

	Economic allocation	
	Rape	Sunflower
Oil Extraction	73.01%	75.11%
Oil Refining	98.19%	98.19%

4.1.4. **Allocation methodology: Mass**

The factors for mass allocation were determined through the proportion of product and by-product produced compared with the total for each relevant process. For the extraction stage, this is shown by equation 4.1-2, which yielded values as shown in table 4.1-3.

$$\frac{\text{Mass of oil produced (1 t)}}{\text{Mass of oil produced (1 t) + Mass of meal (1.5 t)}} = 0.4 \qquad \textbf{Eq: 4.1-2}$$

Table 4.1-3: Mass allocation values used within rape and sunflowerseed oil LCAs

	Mass allocation	
	Rape	Sunflower
Oil Extraction	40%	40.%
Oil Refining	96.45%	96.45%

4.1.5. **System expansion methodology**

The co-products produced in each system are conventionally used as animal feed and it is anticipated that this will also be the case for the residue from the oil-body extraction process. To expand the system to account for this, it was necessary to include a credit for the avoided animal fodder that would not have to be produced due to the production of the system co-products. An appropriate dataset was therefore sought for this purpose.

The Danish LCA Food database contains a Life Cycle Inventory (LCI) for rapeseed meal used as animal feed and this was used for the initial system expansion modelling. It was noted however that this was based on using soy meal and spring barley LCI data, as it was stated that production of rape seed meal would not affect the demand for (and hence production of) rape seed meal since this was governed by demand for the oil (Nielsen, 2003). Whilst this LCI was used for the initial system expansion required, further sensitivity analyses were performed to determine the impact that using LCI data for alternative fodder choices would have on the system. For this purpose, the following datasets from the LCAFood database were used:

- Rapeseed meal (containing soy and spring barley in a ratio of 2.38:1)

- Livestock feed (soy)
- Livestock feed (spring barley)
- Livestock feed (wheat)
- Livestock feed (winter barley)

No system expansion was performed over the cultivation systems as Nemecek et al. (2007), the developers of the cultivation LCI data that was used, stated that within the seed cultivation systems, no straw or beet leaves are harvested for fodder, but treated instead as crop residues.

4.1.6. **Data acquisition**

Specific sources of the data accessed for the foreground system of each LCA are provided in chapters 5 to 8, which outline the LCAs constructed, together with the results generated for each system. Data for the background systems were accessed from the proprietary databases contained within SimaPro, which was the software used for the modelling.

SimaPro contains several different databases, of which the most relevant for the systems developed here were as follows:

- European Life Cycle Database (ILCD) published by the European Commission's Joint Research Centre (JRC)
- LCAFood database, developed as part of a project entitled 'Lifecycle Assessment of Basic Food" undertaken by a consortium of Danish governmental and academic institutes, together with specialist consultants in 2000 - 2003 (www.lcafood.dk, 2013)
- Ecoinvent, which is the world's leading database with consistent and transparent, up-to-date Life Cycle Inventory (LCI) data (www.ecoinvent.org, 2013).

Where multiple datasets were available for a particular material, the suitability of the data was determined based on the age and geographical scope of the data. In addition, sensitivity analyses of the impact of the data in the model were conducted where appropriate.

4.1.7. **Validation**

Validation was performed both for the data accessed and the initial results generated from the LCA model by using comparable data and results presented in peer reviewed journal publications. From the articles outlined in table 3.4-1, several proved particularly relevant for this purpose.

For validation of the data used, one of the articles by Schmidt (2010) was found to be based on his PhD thesis entitled 'Life cycle assessment of rapeseed oil and palm oil' (Schmidt, 2007). Although Schmidt used consequential, rather than attributional LCA, a considerable amount of useful raw data was presented that could be used to validate that used for inventory construction by the rapeseed oil LCA where primary data was not available. In addition, although focusing on rape seed oil as a hydraulic fluid, McManus (2001) presented raw inventory data as part of the PhD thesis that could be used as corroborating much of the data provided by Schmidt.

Fewer corroborating sources were found for the sunflowerseed oil system, with the relevant papers focussing on presentation of results, rather than data. The agricultural handbook FAO/EBRD (1999) provided some useful background information and data here, with Iriarte et al. (2010) also supplying data for validation purposes in the cultivation stage.

The majority of ‘primary’ data used for the seed oil systems was supplied as industry data from Unilever suppliers as will be detailed in section 5. This same data was used for the generation of a series of margarine and butter LCAs published within articles by Shonfield and Dumelin (2005) and Nilsson et al. (2010). The elements of these publications that presented results for the seed oil LCAs could therefore be used to verify that the LCA model developed was generating results in line with those previously calculated. This will be discussed further in chapter 5.

4.2. **SOFTWARE**

With its rigorous calculation methodology, all but the very simplest of LCA systems require an enormous amount of data to be input and manipulated. Although manual analysis is possible with the use of a spread-sheeting package, LCA modelling is generally conducted using one of the many software packages available, most of which have been developed by consultants in the field.

Use of an LCA software package enables all of the required data to be stored in an organised framework and manipulated quickly and easily, enabling repetitive calculations to be performed effortlessly. Such packages usually contain extensive life cycle inventory (LCI) databases which can be accessed readily for building LCA models and allow different life cycle impact assessment (LCIA) methodologies to be used for the analysis. An important facet of any LCA software package is also the ability to access and present results in a structured and clear manner.

Despite the obvious benefits to using an LCA package, there are also several disadvantages which LCA researchers and practitioners must be aware of to ensure that they do not succumb to the potential pitfalls. McManus (2001) describes these as:

- **The black box problem**: Results can be generated very easily and quickly and users may think that the results are accurate and complete when they are not.

- **Not understanding the process**: Untrained people can easily produce "LCAs" without understanding the process, which could lead to inaccurate LCAs being produced.

- **Data quality:** Results can be obtained as soon as any data are put into a database, but this gives no assurance of its usefulness or accuracy.

The European Commission's Joint Research Centre (JRC) publishes a list of all the available software tools within their 'LCA Resources Directory' web pages (www.lca.jrc.ec.europa.eu, 2013). Of the forty plus LCA software tools listed by the directory, many of which cited multiple versions for specific applications, five were identified as having previously been used for oil-seed LCAs through the analysis conducted in chapter 3 and presented in table 3.4-1.

Where the software had been cited in the publication, SimaPro was identified as most frequently used as shown in figure 4.2-1.

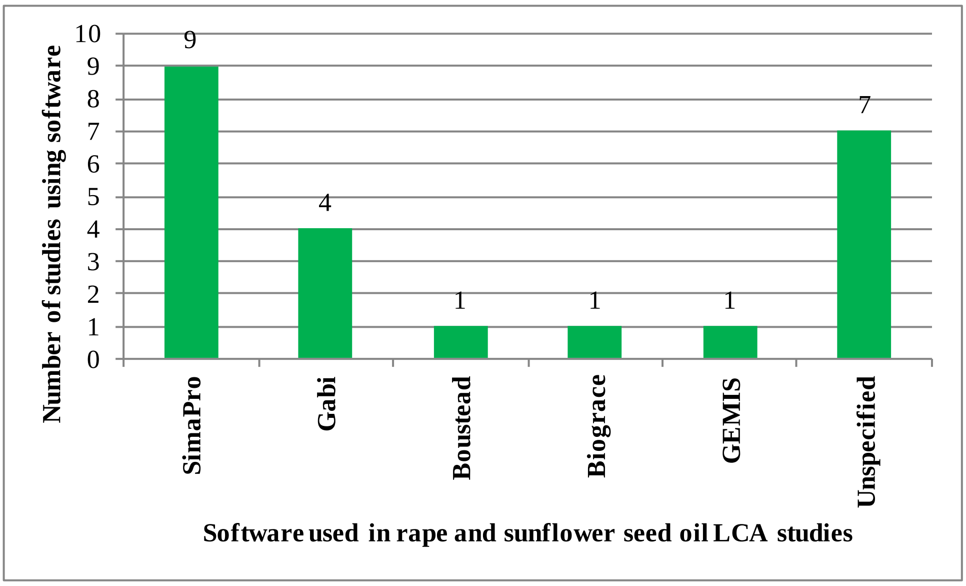

Figure 4.2-1: LCA software programs used for rape and sunflowerseed oil LCAs as identified in table 3.4-1

SimaPro is a proprietary software package for LCA modelling that contains a wide selection of inventory databases as outlined in 4.1.3 and all the major impact assessment methods, which can be edited and augmented to suit the user requirements and modelling scenarios. Given the analysis of past usage, the excellent provision of data supplied within it and considerations above, SimaPro was therefore chosen for the modelling and analysis component of this thesis.

4.3. LCIA METHODOLOGIES

As outlined in chapter 3, to convert the collated inventory of data into meaningful environmental impacts requires the third stage of an LCA, that of LCIA. This is defined by ISO 14040:2006 as the *'phase of life cycle assessment aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts for a product system throughout the life cycle of the product'.*

There are many different LCIA methods available, some of which use midpoint indicators, others that use endpoints and some that have the flexibility of both. The choice of LCIA method is dependent on the analysis and reporting requirements specified as part of the goal and scope.

SimaPro contains a large number of standard impact assessment methods, including those commonly used in Europe, the United States and several single issue methods. From the review of previous LCA studies conducted on rape and sunflowerseed oil systems presented in table 3.4-1, the past use of each of these LCIA methodologies (LCIAM) was assessed. Figure 4.3-1 provides a graphical overview of this data from which it is evident that the two most commonly used LCIAM were CML and Eco-indicator-99 (EI-99). This supports the observation by Goedkoop et al. (2013) who state that CML and Eco-indicator are widely accepted methodologies. From this analysis, together with an assessment of the impact categories required to conform to the goal and scope specification, the choice of LCIAM for this project lay between CML and EI-99.

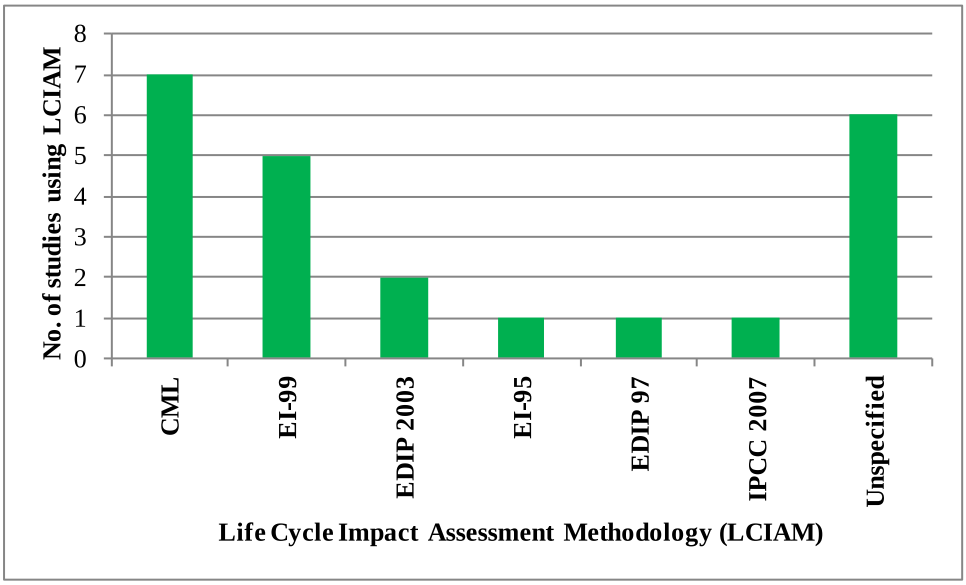

Figure 4.3-1: Use of different LCIA methodologies in previous rape and sunflowerseed oil LCA studies.

Goedkoop et al. (2013) go on to state that the two LCIAM *'are based on different points of departure'*. The CML LCIAM developed by the Institute of Environmental Sciences (Centrum Milieukunde) of Leiden University uses a midpoint approach to impact assessment, whereas EI-99 (and its predecessor EI-95) developed by Pré Consultants B.V, uses endpoints.

Recognising the merits of both approaches, a collaborative project had taken place between 2001 and 2008 involving the Dutch National Institute For Public Health and the Environment (RIVM), Radboud University Nijmegen, CML and Pré, aimed at harmonising these two approaches (ibid). A new LCIAM was developed, named ReCiPe (2008) which as an acronym, represented the initials of the main contributors and collaborators to the project.

ReCiPe(2008) was chosen as the impact assessment methodology for this research as it was borne out of the two most commonly used LCIAMs for past seed oil LCAs, provided the most current methodology and allowed flexibility to model at both endpoint and midpoint levels.

4.4. **RECIPE (2008)**

4.4.1. **Overview of impact assessment method**

ReCiPe comprises eighteen impact categories that are characterised at the midpoint level, most of which are then converted and aggregated into 3 damage categories, or areas of protection (AoP); Human Health, Ecosystem diversity and Resource availability. Figure 4.3-2 provides a simplified overview of these categories, their abbreviations and their linkages to the respective endpoints.

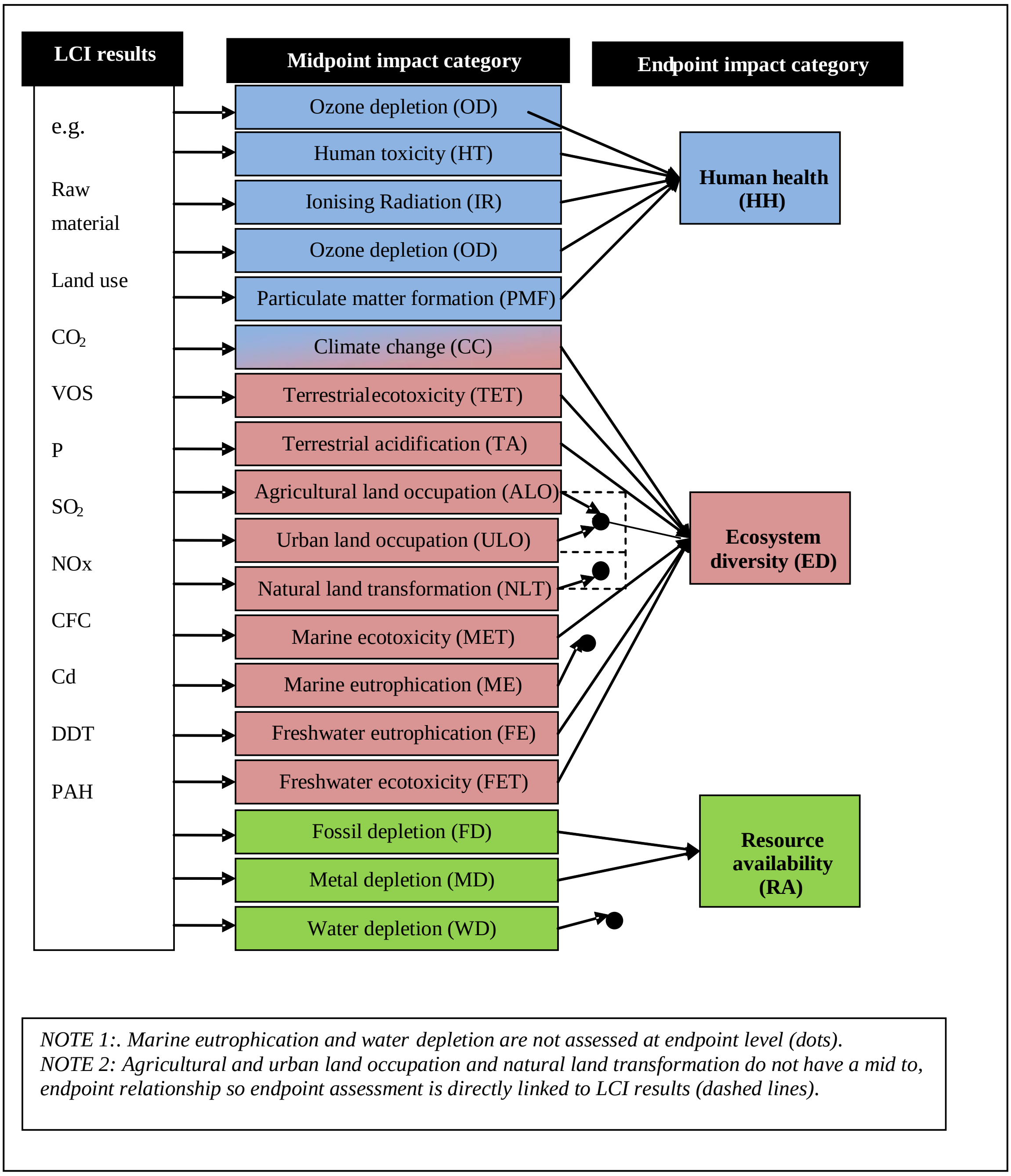

Figure 4.4-1: Simplified overview of ReCiPe LCIA method showing impact pathway and relationship between midpoints and endpoints. *Graphic adapted from Van Hoof et al (2013) and Goedkoop et al (2013)*

Whilst some of the impact categories are known to have higher levels of uncertainty, as will be discussed in the following sections, all of the eighteen categories were used for the analysis outlined in this thesis to enable a complete profile to be constructed.

4.4.2. **Approach to uncertainty**

The ReCiPe (2008) methodology is detailed by Goedkoop et al. (2013) who admit that in common with all LCIA methods, the characterisation models are a source of uncertainty since the relationships are modelled with incomplete and uncertain knowledge of the environmental mechanisms involved. Where these uncertainties are most prevalent, the conversion and aggregation steps are developed based on three different versions of ReCiPe, developed using different cultural perspectives, in line with 'Cultural Theory' by Thompson (1990) (ibid).

The approaches of the three different perspectives are summarised as follows (Goedkoop et al. 2013):

- "Individualist" (I) is based on the short-term interest, impact types that are undisputed, technological optimism as regards human adaptation
- "Hierarchist" (H) is based on the most common policy principles with regards to time-frame and other issues.
- "Egalitarian" (E) is the most precautionary perspective, taking into account the longest time.

The effect of this is that the results generated for each perspective reflect how that archetype would rank the importance of the midpoints within the different damage categories. For this study, the "hierarchist" view was used, for which Goedkoop et al. (2004) state that "*in general, value choices made in the hierarchist version are scientifically and politically accepted"*. An overview of typical societal values for the three archetypes is shown in table B4-2 in appendix B, with the connection between mid and endpoints for the three perspectives shown in table B4.3.

The use of the hierarchist version of ReCiPe led to the GHG characterisations being performed using a 100 year time horizon, whereas the same analysis using the 'I' version would have used the 20 year time horizon and the 'E' version would have utilised the characterisation data for a 500 year horizon. Since the characterisation model used by ReCiPe for GHG estimation was based on the Intergovernmental Panel on Climate Change (IPCC) factors, the use of the 100 year time horizon conformed to the requirements of PAS 2050:2011allowing the data acquired for the climate change midpoint to be used as the CFP.

4.4.3. **Outline of impact categories**

Goedkoop et al. (2013) provide full details of the characterisation models, environmental mechanisms and supporting literature for all mid and endpoint impact categories, as part of the report for ReCiPe (2008). For completeness in this thesis however, a brief description of each the impact categories is provided here.

4.4.3.1. ***Climate Change***

Climate change, as a result of GHG emissions, causes a number of environmental mechanisms that affect both the human health and ecosystem health endpoints. Climate change models are generally developed to assess the future environmental impact of different policy scenarios. In

ReCiPe the IPCC equivalency factors are used for development of the midpoint characterisation factors, with the marginal effect of adding a relatively small amount of GHGs modelled used for endpoint characterisation. The characterisation factor unit is yr/ kg CO_2 eq (Goedkoop et al., 2010).

4.4.3.2. ***Ozone Depletion***

Whilst Goedkoop et al., (2013) note that Hayashi et al., (2006) used a damage function for LCIA to account for ozone depletion impacts on human health, ecosystems and social assets, ReCiPe only characterises damage to human health as uncertainty regarding the other areas was considered too great (ibid). The characterisation factor accounts for the destruction of stratospheric ozone layer by ozone depleting substances (ODS) by evaluating the fate of a marginal increase in ODSs and the resultant increase in UVB exposure. The characterisation factor unit is yr / kg CFC-11. For damage characterisation, factors such as skin colour and cultural habits e.g. clothing are accounted for (ibid).

4.4.3.3. ***Terrestrial Acidification***

Almost all plant species thrive within an optimum level of acidity (Goedkoop et al., 2013) and this acidity is affected by the atmospheric deposition of inorganic substances, such oxides of nitrogen (NOx), phosphates and sulphates, which can cause a shift .in species occurrence (ibid). Midpoint characterisation uses a fate factor based on acid deposition and the resultant changes to soil saturation. Endpoint characterisation then uses an effect factor to model the impact on ecosystems.

4.4.3.4. ***Eutrophication***

Freshwater and marine eutrophication can be defined as nutrient enrichment of the aquatic environment which alters the yield of aquatic biomass such as phytoplankton (algae). Eutrophication in inland waters as a result of human activities is one of the major factors that determine its ecological quality(Goedkoop et al., 2013). ReCiPe uses the integrated assessment model CARMEN (CAuse effect Relation Model) to calculate the change in aquatic nutrient loads, with the limiting nutrients as nitrogen in coastal waters and phosphorous in inland waters. The characterisation factor units are yr/ kg N or P equivalents.

Whist marine eutrophication is modelled only at the midpoint level, ReCiPe uses a damage factor for phosphorous to yield endpoint characterisation for freshwater eutrophication within the eco-systems category.

4.4.3.5. ***Photochemical Oxidant Formation***

Photochemical reactions of NOx and Non Methane Volatile Organic Compounds (NMVOCs) create ozone, which causes a health hazard to humans through inflammation of airways and lungs. This is modelled through using a fate factor which accounts for the marginal change in intake rate of the pollutant due to the marginal change in the emission of the pollutant. This is then combined with an effect factor to generate the endpoint impact within the human health damage category. The characterisation factor unit is yr / kg NMVOC.

4.4.3.6. ***Particulate Matter Formation***

Fine particulate matter with a diameter of less than 10 μm (PM_{10}) is derived from a complex mixture of organic and inorganic substances. PM_{10} causes health problems as it reaches the upper part of the airways and lungs when inhaled. Secondary PM_{10} aerosols are formed in air from emissions from such substances as sulphur dioxide (SO_2), ammonia (NH_3), and nitrogen oxides (NOx) (World Health Organization, 2003), with inhalation of different particulate sizes causing different health problems. ReCiPe models the marginal change in intake rate of pollutant due to the marginal change in emission of the pollutant. The characterisation factor unit is yr/ kg PM_{10} eq.

4.4.3.7. ***Toxicity***

The characterisation factor for human toxicity, freshwater, marine, and terrestrial eco-toxicity accounts for the environmental persistence (fate) and accumulation in the human food chain (exposure), and toxicity (effect) of a chemical (Goedkoop et al., 2013). The chemical 1,4-dichlorobenzene (1,4 DB) was used as a reference substance in the midpoint calculations (to urban air for human toxicity, to freshwater for freshwater eco-toxicity, to seawater for marine eco-toxicity and to industrial soil for terrestrial eco-toxicity). The characterisation factor unit is yr/ kg 1,4 DB.

Fate and exposure factors were calculated by means of ‘evaluative’ multimedia fate and exposure models, while effect factors were derived from toxicity data on human beings and laboratory animals. The commonly applied multimedia fate, exposure and effects model, USES-LCA (Uniform System for the Evaluation of Substances) was adapted for use within ReCiPe (ibid).

4.4.3.8. ***Ionising Radiation***

Human health is damaged through exposure to routine releases of radioactive material to the environment. Within ReCiPe the midpoint is chosen at the level of exposure in Becquerel (Bq), with one Becquerel equivalent to one decay per second. (Goedkoop et al., 2013). Within the damage analysis, the carcinogenic and hereditary effects are then used to perform effect and damage analysis to develop the human health impact endpoints. The characterisation factor unit is yr/kg Uranium 235 (U_{235}) eq.

4.4.3.9. ***Land Use***

ReCiPe concentrates the damage to ecosystems due to the effect of land use in terms of both occupation and transformation, identifying three different types of land at the midpoint level; 'agricultural', 'urban' and 'natural'. The characterisation factor units are all in area occupied over a year (m^2*yr). ISO 14001:2004 defines impacts as *'any change to the environment, whether adverse or beneficial, wholly or partially resulting from an organisation's aspects'*. As such it could be perceived that land occupation midpoints are not actually impacts at all, but merely a statement of area covered from which impacts can arise.

Characterisation at the endpoint level models impacts arising from land use by taking the occupation of a certain area of land during a certain time, and the transformation of that certain area of land (Goedkoop et al., 2013) to model the potential disappeared fraction of species (PDF) (ibid).

4.4.3.10. ***Water Depletion***

Water is a scarce resource in many parts of the world, but also a very abundant resource in other parts of the world. Extracting water in a dry area can cause considerable damage to ecosystems and human health, but so far no models are available to express the damage on the endpoint level. ReCiPe includes a midpoint indicator that simply expresses the total volume of water used in m^3 (Goedkoop et al., 2013).

4.4.3.11. ***Metal Depletion***

Metals are extracted from minerals, which most often entail mining for the deposits which contain several different mineral resources. The minerals or metals become the economic output of a mining operation and are therefore also called commodities (Goedkoop et al., 2013). The characterisation factor unit is kg Iron (Fe) equivalents. In the description of the area of protection, the damage is defined as the additional costs to society as a result of cost increases for the metal in question resulting from extraction. This cost is calculated by multiplying the marginal cost increase of a resource with an amount that is extracted during a certain period (ibid).

4.4.3.12. ***Fossil Depletion***

Goedkoop et al. 2013 state that the term ‘fossil fuel’ refers to a group of resources that contain hydrocarbons, ranging from volatile materials like methane, to liquid petrol, to non-volatile materials like anthracite coal. The characterisation factor for fossil depletion is based on the projected change in the supply mix between conventional and unconventional oil sources and the unit used is MJ. Unconventional fossil resources are generally more energy intensive and more costly to produce, compared to conventional fuels; therefore production of unconventional fuels is only viable when the overall price level for the fuel is high enough to cover the costs. Using this premise, ReCiPe models this impact category by calculating the marginal cost to society of producing these unconventional fuels to replace the depleted fossil fuels.

4.4.4. **Normalisation**

The ReCiPe reference values for normalisation are based on an LCA study of the Global and European economic systems in the year 2000 as stated in Wegener Sleeswijk et al. (2008). Europe comprised twenty eight European countries, consisting of the twenty five countries of the European Union in 2006 (including Finland) supplemented with Iceland, Norway and Switzerland (referred to as EU25+3).

Van Hoof et al. (2013) outline the normalisation process, which is performed according to equation 4.4-1, with Wegener Sleeswijk et al. (2008) further explaining that this is based on the use of reference interventions in *'units per year'*.

$$NS_{A,j} = \frac{\sum_x CF_{x,j} \text{ x } M_{a,x}}{\sum_x CF_{x,j} \text{ x } M_{NR,x})} \qquad \textbf{Eq: 4.4-1}$$

Where:

$NS_{A,j}$ = Normalised score of product 'A' at the impact category 'j' (dimensionless)

$CF_{x,j}$ = Characterisation factor for LCI flow 'x' in impact category 'j' (kg-eq/kg)

$M_{a,x}$ = The amount of LCI flow 'x' in product 'A' (kg/yr)

$M_{NR,x}$ = The amount of LCI flow 'x' in the normalisation reference (kg/yr)

It should be noted however that in SimaPro, the ReCiPe normalisation is performed by multiplying the characterised values by the inverse of the normalised values as shown in equation 4.4-2, which is mathematically equivalent to equation 4.4-1.

$$NS_{A,j} = \sum_x CF_{x,j} \text{ x } M_{a,x} \text{ x } \frac{1}{\sum_x CF_{x,j} \text{ x } M_{NR,x})} \qquad \textbf{Eq: 4.4-2}$$

Endpoint normalisation is performed per damage category (Resources, Human Health, Eco-Systems AoPs), whereas midpoint normalisation is carried out per impact. For endpoint normalisation therefore, three normalisation values exist for both the Global and EU25+3 regions, with all of the impacts for a particular AoP normalised against the relevant factor. Midpoint normalisation on the other hand requires a different normalisation value for each impact category. All endpoint and midpoint normalisation parameters can be accessed from Wegener Sleeswijk et al. (2008) or downloaded in an Excel file from www.lcia.net (2013).

Large variations exist between the normalisation parameters for the two regions, with the most noteworthy differences arising for the toxicity impacts, a characteristic that Wegener Sleeswijk (2013) state is most probably a result of the relatively high percentage of world GDP (27%) for which the EU region is responsible in comparison with its population. Whilst the lower differences within the other impact categories may lead this theory to be questioned, they go on to note that '*the European contribution to other impact categories – mostly around 10% or less – is rather low. For the emission-related impact categories, this may be due to the use of cleaner technologies in this region'*.

The uncertainty in normalisation parameters can arise through incomplete emission data, characterisation factors, or both (Heijungs et al., 2007; Kim et al., 2012). Wegener Sleeswijk et al. (2013) state that the differences in global and European normalisation figures for the toxicity categories emphasise the relatively high uncertainty of the normalisation factors for toxicity in both Europe and the world. This sentiment is echoed by Van Hoof et al. (2013) who highlight midpoints that have a lot of contributing elementary flows (e.g. toxicity indicators) as potentially having a higher level of uncertainty through incompleteness in characterisation factors.

Van Hoof et al. (2013) advocate combining the use of normalised endpoint indicators for ranking of significance with characterised midpoint values for reporting results, a technique that was investigated in chapter 5 and used for the remaining analysis.

4.5. **CONCLUDING SUMMARY**

This chapter built on the introductory information concerning LCA that was provided in chapter 3, by detailing some of the methodological considerations and procedures used to fulfil the research objectives of this thesis. The parameters and decisions involved when specifying the goal and scope of the required LCAs were outlined and approaches required for data collection and modelling discussed. Further detailed specification of the required elements of each goal and scope will be discussed in the relevant chapters.

Detail on the chosen LCIA methodology was provided, in addition to an outline of the impact categories which will be most prevalent in the results and discussion for the following chapters. Having completed all introductory and specification stages, the following chapters will detail the analysis conducted and results generated to fulfil the aims of the thesis.

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CHAPTER 5. LCA OF RAPE AND SUNFLOWERSEED OILS

Since the primary aim of the work presented by this thesis was to determine whether the novel processing route for production of food grade edible oil emulsions via aqueous extraction of oil-bodies (OB) had a better environmental profile than that of the existing technology route, it was important to obtain a baseline for comparison. As outlined in chapter 3, whilst a multitude of different Life Cycle Assessment (LCA) studies have been performed over product systems which use rape and sunflowerseed oils as part of the process, only four presented results for the seed oils as products (McManus et al., 2004; Narayanaswamy et al., 2004; Schmidt, 2010b; Roïz and Paquot, 2013). For the research presented here, it was vital to generate an LCA specific to this system, to ensure that all variables and assumptions were known and could be retained as identical in the subsequent analysis of the OB system.

LCA models were therefore created for both rape and sunflowerseed oils, that could be used both as a building block within the overall research and to fulfil the first objective, to use LCA to establish the environmental loads for the production of refined rapeseed and sunflowerseed oils, together with the relative contributions from each of the processing stages. This will provide transparent results for the given set of parameters in addition to the following:

a. Quantification of the impact of using different methods for co-product allocation within the attributional LCA models

b. Identification of the effect of using different normalisation data sets for analysis of significance.

5.1. RELEVANT LITERATURE

In addition to reviewing previous rape and sunflowerseed oil LCA studies to acquire an understanding of the use of methodological parameters, as discussed in chapters 3 and 4, the relevant papers were also scrutinised to identify any that could be used to acquire base data for modelling purposes, or as a source of validation for the results generated.

Most papers focused on presentation of the results of the LCA without detailing the raw inventory data. However Iriarte et al. (2010) contained inventory data for rape and sunflowerseed production in Chile, which although not geographically specific for this study, provided a useful source of cultivation data in addition to the results and commentary.

Whilst Schmidt (2007) and McManus (2001) could both be used for validation purposes, the most useful source of inventory data for both rape and sunflowerseed oils was provided by Nilsson et al. (2010) in their comparison paper of margarines and spreads. SEIBI project consultant E. Dumelin was able to supply background assumptions to the analysis presented in this article via an unpublished internal report. He also confirmed that the same data had been used for the assessments outlined by Shonfield and Dumelin (2005) that provided a breakdown of environmental impact by process stage for the seed oil production system. This provided a direct source of data and comparative results that could be used for construction and validation of the basic model.

5.2. **SYSTEM DEFINITION**

As outlined in chapter 2, the most common processing route for the production of edible oil for food use is the two stage pre-pressing and solvent extraction route, followed by the necessary refining stages. Attributional LCA models of both rapeseed and sunflowerseed oil product systems were constructed using the SimaPro 7 software system with the functional units set at: ‘receipt of 1 tonne of refined oil at food processor.’ The starting boundary was cultivation of seed and the finishing boundary set as delivery of oil to food processor. The simplified process flow used for both product systems was as depicted in figure 5.2-1, with the main process stages being cultivation, extraction and refining.

Rapeseed cultivation and extraction were modelled as taking place in Germany, refining in The Netherlands and receipt at food factory in U.K (Leicester). For the sunflowerseed oil system, the locations were chosen as cultivation and extraction in Spain, refining in The Netherlands and receipt at food factory in UK

Whilst this indicates a relatively simple flowsheet, creation of the LCA entailed each input stream being further expanded to develop a list of inputs and outputs for each individual system. The result when modelled in SimaPro was a complex process network involving over 2000 process nodes (input values) as shown in figure 5.2-2. Note that for ease of display, only those inputs (process nodes) with a contribution of 5% or higher are shown. This simplifies the network such that only 13 out of the possible 2053 nodes are displayed.

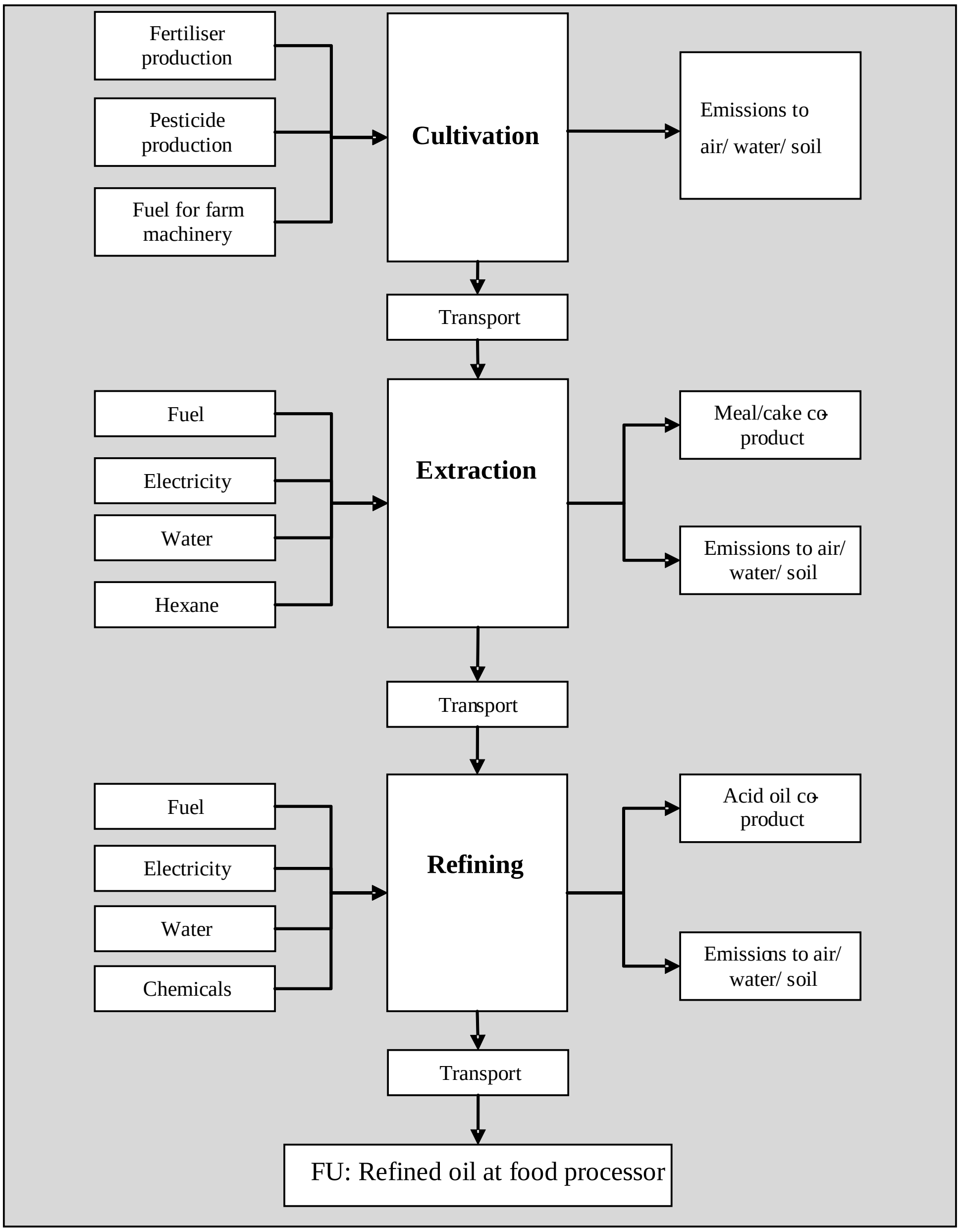

Figure 5.2-1: Simplified flow-sheet of seed oil production process

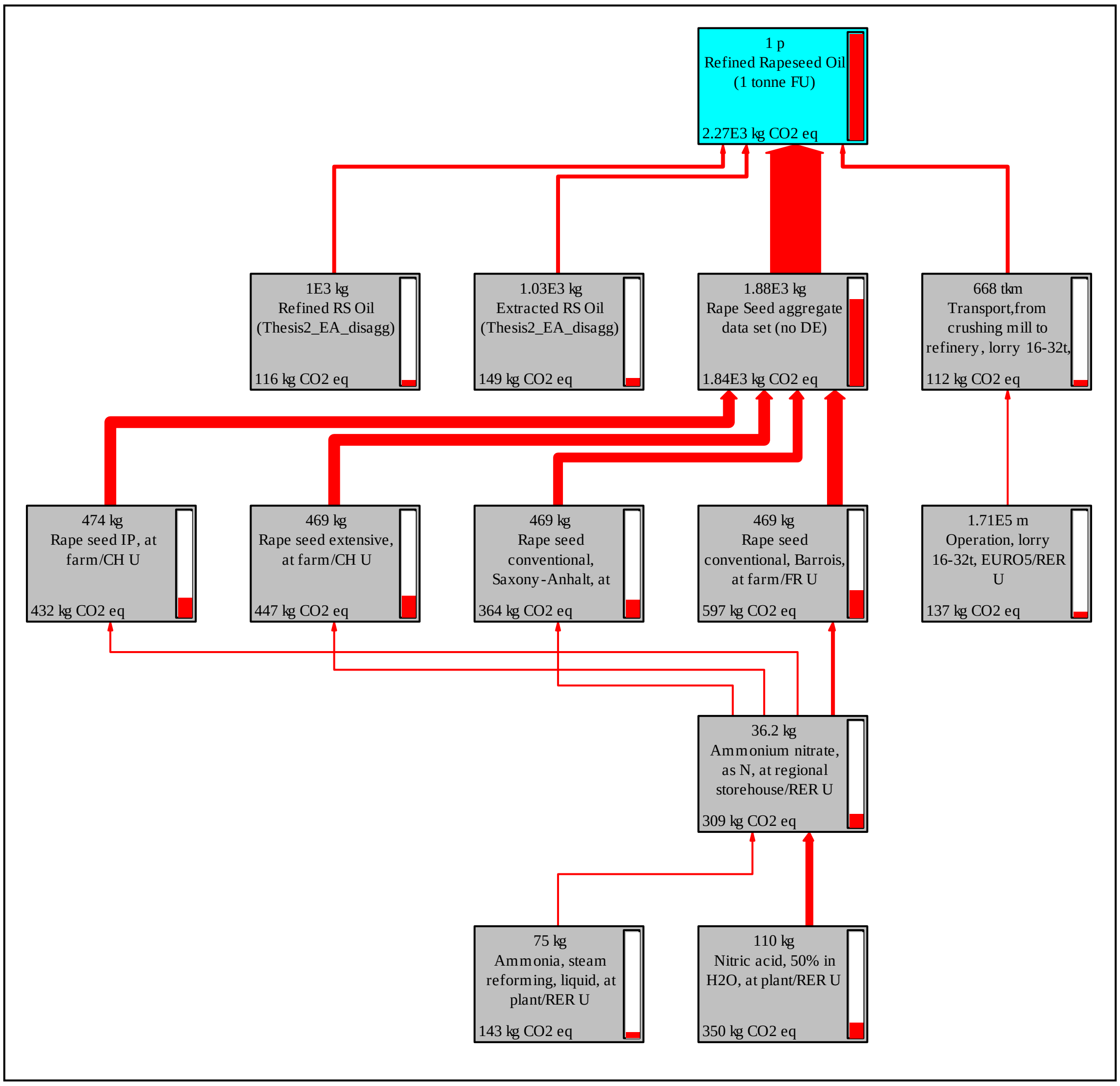

Figure 5.2-2: SimaPro network for refined rapeseed oil at 5% cut-off

5.3. DATA GATHERING

The initial step was to acquire data for each of the foreground processes; agriculture, crude oil extraction, refined oil production and transport modes and distances. Nilsson et al.(2010) present an LCI for the production of various seed oils, including rape and sunflower, which was predominantly taken from Unilever manufacturing sites and suppliers. SEIBI project consultant E.Dumelin, was able to supply the original (unpublished) internal report upon which this and a previous paper (Shonfield and Dumelin, 2005) were based, which enabled all assumptions to be reviewed and additional process inputs to be acquired. This data was also compared against that from literature sources (McManus, 2001; Schmidt, 2007) and data avilable from the Ecoinvent database as supplied within SimaPro as will be discussed in the following sub sections.

Data for the background processes required for each of the inputs such as electricity, fertiliser production, diesel production and steam generation was taken from the EcoInvent 2.2 database supplied in SimaPro, for the geographical area of the process in question e.g. for generation of electricity used in Rapeseed oil extraction, the Germany power mix was utilised. For the system expansion sensitivity analyses, the data was utilised from the LCAFood database supplied in SimaPro.

5.3.1. **Cultivation**

The cultivation data presented by Nilsson (2010) contained an aggregate amount for fuels consumption of farm machinery, together with amounts of fertiliser and pesticide used, based on rape seed cultivation in Germany and sunflowerseed cultivation in South Africa.

SimaPro contained seven different Ecoinvent datasets concerning cultivation of rapeseed and two for sunflowerseed, in different geographical locations and using different technology. Following a review of each of these datasets, it was clear that each EcoInvent dataset contained considerably more detail than that afforded by the Nilsson data and were equally as current. It was therefore decided to utilise one of the Ecoinvent processes to model the cultivation stage in place of the Nilsson (2010) data. Details of the datasets concerned are given in table 5.3-1 for which Nemecek et al. (2007) provide the following descriptions of the technology specifications:

1. *Conventional production* in the Swiss context is agricultural production complying with legislation but not meeting the minimum requirements for integrated production
2. *Integrated production* refers to agriculture meeting the ecological requirements defined by the *Direktzahlungsverordnung* decree (direct payments regulation – part of Swiss agricultural policy)
3. *Organic production* complies with the requirements for organic production. Application of synthetic pesticides and fast-acting mineral fertilisers is not permitted.

The individual datasets contained considerable differences with regard to fertiliser and pesticide usages, both in terms of quantities and types used. From reviewing the reference material for all rape and sunflowerseed datasets (Jungbluth et al.,2007; Nemecek et al., 2007) it was apparent that this was due to regional differences introduced through fertiliser and pesticide application recommendations and findings of local experts.

Table 5.3-1: Ecoinvent rape and sunflowerseed cultivation datasets within SimaPro

Name of LCI dataset	Description	Reference
Rape seed conventional, at farm/DE	Location: Germany Technology: Conventional production	Jungbluth et al. (2007)
Rape seed conventional, Saxony-Anhalt, at farm/DE	Location: Saxony-Anhalt, Germany Technology: Conventional production	Nemecek et al. (2007)
Rape seed conventional, Barrois, at farm/FR	Location: Barrois (France) Technology: Conventional production	Nemecek et al. (2007)
Rape seed extensive, at farm/CH	Location: Swiss Lowlands Technology: Integrated production with extensive plant protection	Nemecek et al. (2007)
Rape seed IP, at farm/CH	Location: Swiss Lowlands Technology: Integrated production	Nemecek et al. (2007)
Rape seed, at farm/US	Location: USA Technology: Conventional production	Nemecek et al. (2007)
Rape seed, organic, at farm/CH	Location: Swiss Lowlands Technology: Intensive organic production	Jungdbluth et al. (2007)
Sunflower conventional, Castilla-y-Leon, at farm/ES	Location: Spain Technology: Conventional production	Nemecek et al. (2007)
Sunflower IP, at farm/CH	Location: Swiss Lowlands Technology: Conventional production	Nemecek et al. (2007)

In order to determine whether the differences observed would have an impact on the resultant LCA model, a comparative analysis of the datasets was performed using ReCiPe (2008) LCIA, to determine the levels of difference introduced by the different agrochemicals. The four largest rapeseed producers by order of importance are the EU, China and India (Rosillo Calle, 2009). The rapeseed dataset based on US cultivation was therefore excluded from the analysis as potentially unrepresentative. In addition it was felt that seed from organic cultivation would most likely be used for production into bottled oil finished product, rather than processed for food industry use; therefore the organic dataset was also excluded.

As anticipated, the different input levels of agrochemicals had a considerable impact on the results obtained for each cultivation model, with a particularly wide variation evident within the rapeseed cultivation datasets. The normalised midpoint values for each of the five rapeseed datasets analysed, are shown, numbered 1 to 5, in figure 5.3-1. This figure also shows the result for an aggregate dataset which will be discussed later in this section. The raw characterised data for this can be found in table C5.1 in appendix C.

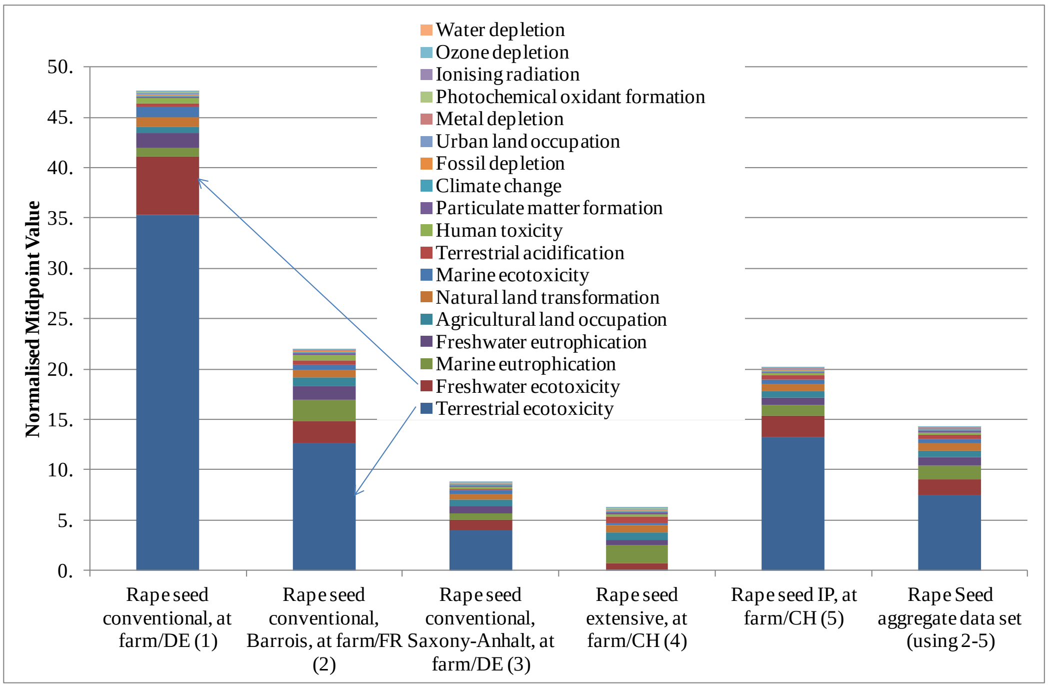

Figure 5.3-1: Comparison of rapeseed cultivation datasets – normalised midpoint data: Impacts shown in order of increasing relative importance

From this it was evident that the largest impacts were derived from the terrestrial eco-toxicity (TET) category for all of the cultivation models apart from the Swiss model for integrated production with extensive plant protection, which contained no fungicides, insecticides or growth regulators. The TET impact for the 'Rapeseed conventional at farm/ DE U', the generic German cultivation dataset was over twice the value for any other data sets.

All of the TET impacts with a contribution of 0.1% or over were as a result of four agrochemicals: Carbendazim fungicide, Metazachlor herbicide, Phosphorous Fertiliser and Cypermethrin pesticide. Of these, the synthetic pyrethroid pesticide Cypermethrin, which Mulligan et al. (2006) note is '*the most widely used pesticide in UK oilseed rape*,' was by far the largest single contributor ranging from 82.5% contribution for the extensive Swiss cultivation to 99.8% for the generic German cultivation.

Although not visible from the normalised data presented in 5.3-1, the differences in input/output data caused considerable variability across the full range of impacts. For example with the characterised data for climate change for example ranging from 0.776 tonne CO_2eq (tCO_2eq) for the German Saxony-Anhalt data, to 1.326 tCO_2eq for the generic German results. Such variability can also be found in the results reported in literature for cultivation of rapeseed, with Iriarte et al. (2010) citing the climate change impacts for rapeseed cultivation as 0.82 t CO_2eq per tonne of seed, whereas Twining and Clarke (2009) report 1.5 tCO_2eq for the same FU.

In reality, the seed for production of the oil would not be sourced from a single supplier / geographical location, but would more likely be sourced from a range of suppliers / cultivators (Towle, 2010). To model this most effectively therefore, an aggregate dataset was developed to supply the cultivation data for the LCA model, assuming that equal portions came from the different locations. However, since substantive research had failed to reveal the reason that the Cypermethrin application quantities in the generic German dataset were almost five times the amount specified for application within the other German dataset (Saxony-Anhalt) and almost twice that input in the Barrois and Swiss integrated production models, it was decided to exclude this dataset from the aggregate due to mis-alignment with the others.

Whilst there were only two available sunflowerseed cultivation datasets, based in Spain and Switzerland, these again exhibited considerably different environmental burdens, as shown in figure 5.3-2. The largest impact category for sunflower cultivation was marine eutrophication (ME) which was traced back to the nitrate emissions to water, due to the use of the herbicide Linuron, a finding that was supported by Iriarte et al. (2013) who examined the cultivation of sunflower and rape seed in Chile. ReCiPe characterises all nitrate emissions within the ME category as described in Chapter 4.

As with the rape seed, sunflowerseeds would in practice be acquired from multiple geographical locations for crushing at the oil mill and it was therefore decided to generate an aggregate data set which comprised equally quantities of the two data sets. These aggregate datasets were used within all modelling that follows.

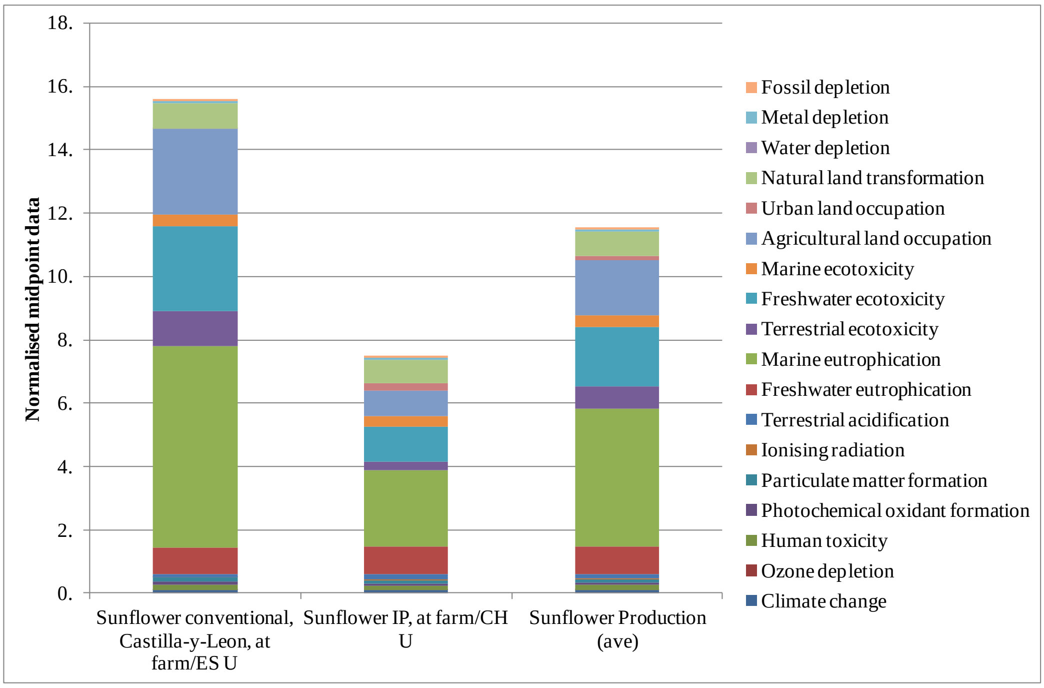

Figure 5.3-2: Sunflowerseed cultivation datasets – normalised midpoint data

5.3.2. **Extraction and refining**

Following the analysis of the Nilsson data it was clear that for extraction and refining of the oil, the data presented was consistent with that supplied in the other sources reviewed and therefore appropriate for use in this model. This is presented in table 5.3-2.

Table 5.3-2: Table of inventory inputs for rape and sunflowerseed oil systems. Source: Nilsson et al. (2010), adapted with data from Dumelin (2010)

		INPUTS		OUTPUTS	
	Units	**Rape seed oil**	**Sunflowers eed oil**	**Rape seed oil**	**Sunflower seed oil**
Oil extraction					
Crop input to crushing mill	kg/tonne	2500	2500		
Meal production	kg/tonne			1500	1500
Crude oil production	kg/tonne			1000	1000
Electricity	MJ/tonne	500	500		
Hexane (to account for loss)	kg/tonne	2	2		
Steam	MJ/tonne	1680	1680		
Refined oil production					
Crude oil input	kg/tonne	1046.46	1046.84		
Acid oil co-product	kg/tonne			36.85	37.95
Refined oil product	kg/tonne			1000	1000
Activated carbon	kg/tonne	2.02	5.05		
Bleaching earth	kg/tonne	7.06	3.03		
Electricity	kWh/tonne	54.79	54.8		
Diesel Fuel	kg/tonne	8.02	8.02		
Steam	kg/tonne	265.91	266.01		
Process water	m^3/tonne	0.16	0.16		
Cooling water	m^3/tonne	7.12	7.12		
Nitrogen	kg/tonne	6.305	5.04		
COD	kg/tonne			0.2548	0.2548

5.3.3. **Transportation**

Transportation distances had been specified by the Nilsson article for rapeseed cultivation and extraction in Germany and Refining in the Netherlands, together with sunflowerseed cultivation and extraction in South Africa and refining in The Netherlands. These are shown in table 5.3-3. To ensure that the transport distances used in the model were consistent with the geographical specificity of the cultivation datasets however, it was necessary to modify some of these values.

Table 5.3-3: Table of transportation distances for rape and sunflowerseed systems

Transport	**Units**	**Rapeseed**	**Sunflower seed**	**Rapeseed**	**Sunflowerseed**
		Source: Nilsson et al. (2010)			
Transport farm to mill (road)	km	65	100	65	250
Transport mill to refiner (sea)	km		12300		1756
Transport mill to refiner (road)	km	650		650	88
Transport refiner to factory (sea)	km			390	390
Transport refiner to factory (road)	km	50	50	196	196

Since aggregate datasets covering multiple geographic locations were used for the cultivation stage, the transportation requirements would clearly be different for different locations. Rather than develop an average transport dataset, it was decided to use one of the cultivation locations as a basis and perform a sensitivity analysis on the impact of transportation within the model. The LCA model was thus specified as: rapeseed cultivation and extraction in Germany, refining in The Netherlands and receipt at food factory in U.K (Leicester). For the sunflowerseed oil system, the locations were chosen as cultivation and extraction in Spain, refining in The Netherlands and receipt at food factory in UK

5.4. **RESULTS AND DISCUSSION**

The LCA models created using the methodology outlined in the previous section were firstly validated by comparison of results obtained using the input data from Nilsson et al. (2010) with those reported by Shonfield and Dumelin (2005), which had previously used this data. Following successful validation, analysis was performed to provide the following information.

1. The environmental loads of rape and sunflowerseed oil production
2. The relative contributions of each of the individual processing stages
3. Sensitivity analysis of alternative methods for treatment of co-products
4. The impact of using different normalisation sets
5. The sensitivity of the transportation assumptions.

Overview results data is presented within this chapter, with further detailed information available in Appendix C.

5.4.1. **Validation of model against literature**

As stated previously, the data utilised for the SimaPro models was acquired from Nilsson et al. (2010) who utilised the same input data as Shonfield and Dumelin (2005) (Dumelin, 2010). Since Shonfield and Dumelin (2005) reported results for the relative contributions of each process stage, this data was used as a comparator with the data generated by the SimaPro models to validate that the SimaPro model was functioning correctly. However, as detailed in 5.3.1, the models constructed for the research outlined by this thesis use ecoinvent cultivation datasets in preference to the cultivation data of Shonfield and Dumelin (2005). Therefore, for the purpose of validation, the Shonfield and Dumelin cultivation data was used.

Figure 5.4-1 shows the relative contributions of extraction, refining and transport to the single-score value generated using the SimaPro models with LCIA performed using ReCiPe(2008) endpoints. This indicates the correlation of data output from these LCA models vs those generated by Shonfield and Dumelin (2005) which are labelled as ‘Shonfield seed data.’

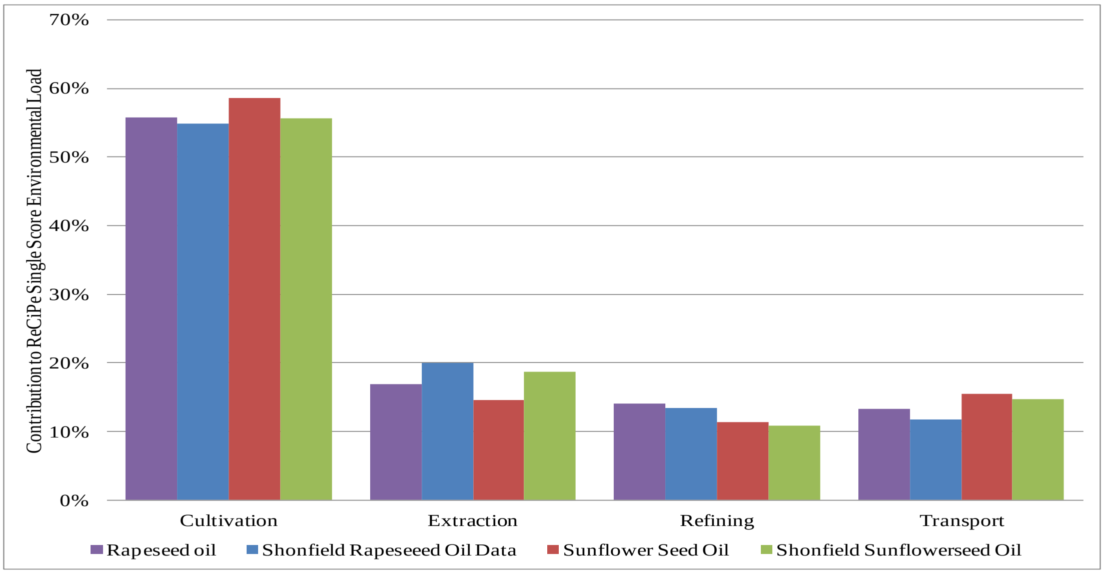

Figure 5.4-1: Comparison of Shonfield and Dumelin (2005) data with current SimaPro model

Whilst an exact match would not be expected due to discrepancies in data sets for the background processes and the use of a different (unspecified) LCIA methodology, the values obtained with the SimaPro models are largely consistent with the Shonfield and Dumelin data, providing confidence in the models generated. It should be noted however that this data is purely for validation purposes, since the more comprehensive cultivation datasets were used in the main analysis as will be reported in the following sections.

5.4.2. **Environmental loads of refined rape and sunflowerseed oil production**

Normalised analysis of the systems at the endpoint level, using European normalisation values are shown in figure 5.4-2, where it can be seen that both product systems have the greatest impact in the Damage to EcoSystems area of protection (AoP), with the results for the Damage to Resources and Damage to Human Health AoPs being almost equal.

In all cases, the impacts for the sunflowerseed oil system are greater than those for the rapeseed oil system, with the results in the ecosystems category showing the largest difference, driven by the agricultural land transformation impacts for that endpoint. This result is consistent with Shonfield and Dumelin (2005) in which they comment that *'Sunflower oil tends to have high environmental impacts because of the relatively low yields per hectare compared to other crops.'*

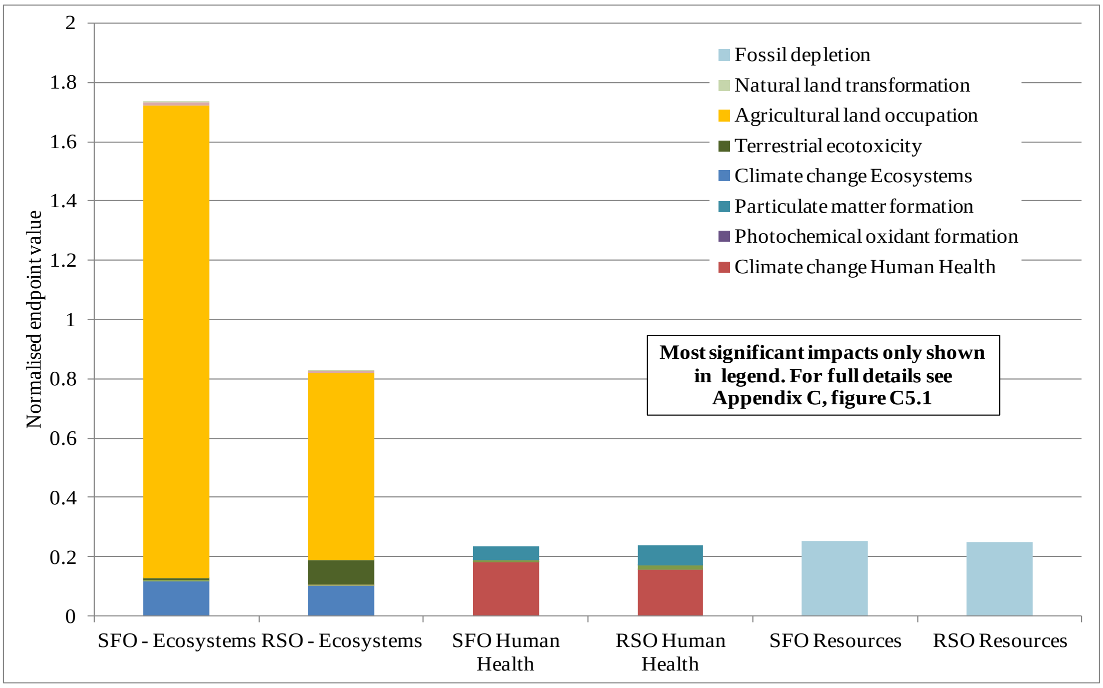

Figure 5.4-2: Normalised endpoint results for analysis of rape and sunflowerseed oil LCA (ReCiPe(2008)). RSO = Rapeseed oil; SFO = Sunflowerseed oil

Within the Resources AoP, the fossil depletion impacts dominated the result, contributing 99.9% of the normalised impacts for both types of seed oil. This was principally driven by the contribution from the agro-chemicals production required for cultivation of the seed. For the Human Health AoP, the impacts attributed to climate change were the largest for both seed oils, delivering 65.9% of the normalised impact for the rapeseed and 76.6% for the sunflowerseed oil system. Particulate matter formation was the second largest contributor here, with 28.7% and 19.8% of the normalised endpoint impacts for rape and sunflowerseed oils respectively. Again, these impact results arose principally from the cultivation stage of the process. The dominant nature of cultivation will be discussed further in the next section.

Assessing both systems using midpoints, the characterised impact results at the midpoint level were as shown in table 5.4-1, from which the top five impacts as identified through midpoint normalisation have been indicated with their ranking in brackets, after the raw impact figure.

Table 5.4-1: Characterised midpoint impacts of rape and sunflowerseed oils

Impact category	Unit	Rapeseed Oil	Sunflowerseed Oil
Climate change	kg CO_2 eq	2271	2597
Ozone depletion	kg CFC-11 eq	0.0002	0.0002
Human toxicity	kg 1,4-DB eq	371	238
Photochemical oxidant formation	kg NMVOC	8.5	9.5
Particulate matter formation	kg PM_{10} eq	5.3	3.6
Ionising radiation	kg U_{235} eq	203	246
Terrestrial acidification	kg SO_2 eq	26.8	10.5
Freshwater eutrophication	kg P eq	0.7 (5)	0.8 (5)
Marine eutrophication	kg N eq	26.3 (3)	86 (1)
Terrestrial ecotoxicity	kg 1,4-DB eq	115 (1)	10.9
Freshwater ecotoxicity	kg 1,4-DB eq	31.5 (2)	40.6 (2)
Marine ecotoxicity	kg 1,4-DB eq	8.0	7.5
Agricultural land occupation	m^2a	6003	15158 (3)
Urban land occupation	m^2a	57.6	97.1
Natural land transformation	m^2	0.3 (4)	0.4 (4)
Water depletion	m^3	7.4	6.2
Metal depletion	kg Fe eq	98.7	99
Fossil depletion	kg oil eq	418	424
N.B.	The ranking of the top five impacts as identified through midpoint normalisation is indicated in brackets		

Midpoint normalisation indicated that toxicity and eutrophication impacts were the most noteworthy for both seed oils, differing from the endpoint normalisation results presented in 5.4-2, which determined agricultural land occupation, fossil depletion and climate change as more prominent in both systems. Midpoint normalisation can be subject to higher levels of uncertainty than endpoint normalisation as discussed in chapter 4, with Van Hoof et al. (2013) and Wegener Sleeswijk et al. (2000) stating that the highest levels of uncertainty through incomplete emissions and characterisation factors arise within toxicity midpoint normalisation.

As ReCiPe (2008) uses the IPCC (2007) equivalence factors as the GWP characterisation method (Goedkoop et al., 2013) and the scope of the assessment conforms to PAS2050:2011, the climate change midpoint impact category results provide the CFPs for the two systems. From table 5.4-1 it can be seen that the CFP of each tonne of refined rapeseed oil delivered to the food factory is therefore 2.3 tCO_2eq and the corresponding CFP for the sunflowerseed oil system is 2.6 t CO_2eq per tonne of refined oil delivered to food processor.

Shonfield and Dumelin (2005), comment that *'Sunflower oil tends to have high environmental impacts because of the relatively low yields per hectare compared to other crops*. The reduced yield leads to higher impacts per tonne to be derived through fertiliser use, with the greenhouse gas (GHG) impacts resulting from nitrous oxide (N_2O) liberation being the predominant cause of the increased CFP of sunflowerseed oil compared with its rapeseed oil counterpart.

5.4.3. **Relative contributions from each of the processing stages**

As indicated in the previous section, the impacts attributed to cultivation stage are the over-riding contributors at both the mid and endpoint level. To examine the relative contributions of the main processing stages more thoroughly, the SimaPro calculation setup was modified to extract data for each of the main stages: cultivation, extraction, refining and an aggregate of all the transportation. The analyses at the endpoint level are given in figure 5.4-3, where the dominant contributions from the cultivation stage can be very clearly seen.

It should be noted that this data also clearly demonstrates the impact of using the more detailed cultivation datasets, since the results presented using the Nilsson et al. (2010) data in figure 5.4-1 indicated contributions to the single score environmental load of 55.8% and 58.6% for rapeseed and sunflowerseed cultivation respectively. The results generated using the aggregate datasets which have been chosen for ongoing analysis, as detailed in 5.3.1 indicate that these contributions are 87.8% for the rapeseed cultivation and 93.0% for sunflowerseed cultivation. Details of this data can be found in table C5-3 in appendix C.

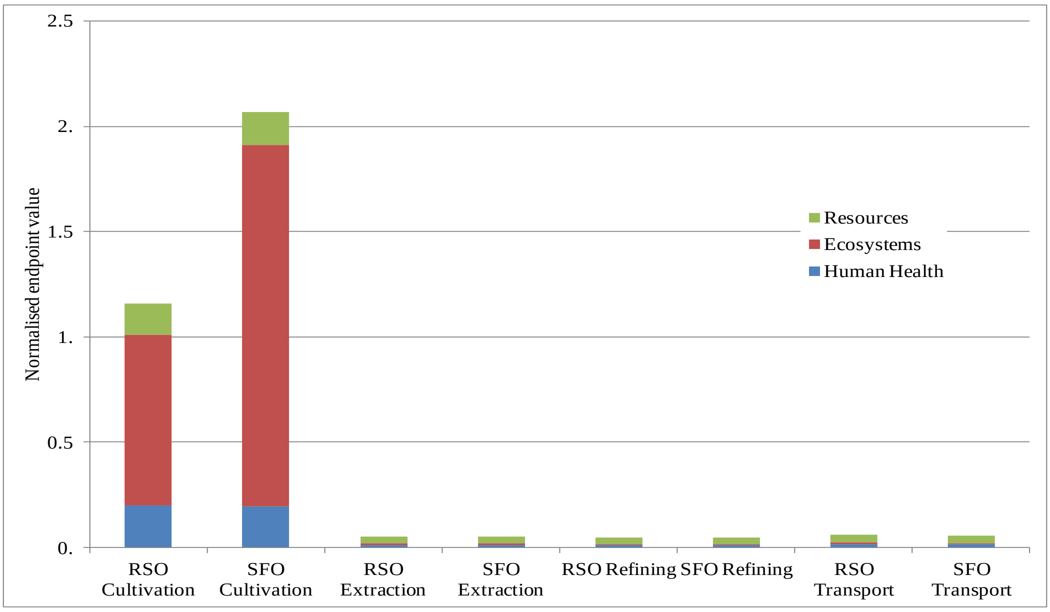

Figure 5.4-3: Relative contribution of processing stages to normalised endpoint

This was also evident from looking at figure 5.4-4, which presents the five impacts with the largest normalised endpoint values for both oils, from which the dominating contribution of cultivation to each of these categories can clearly be seen.

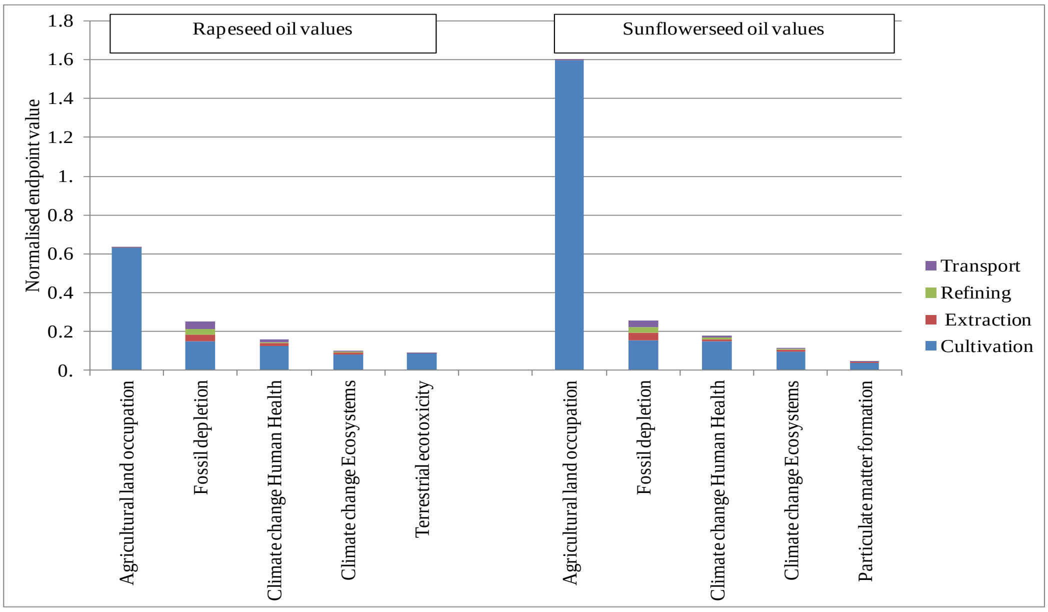

Figure 5.4-4: Most prominent impact categories for oil production systems as indicated by endpoint normalisation

The results of the normalised midpoint analysis are provided in figure 5.4-5 which indicates the relative contributions to each of the five most prominent midpoint impacts for both oils, together with the relative contributions to the climate change impact category, which features in the top five impact categories when assessed using normalised endpoints. The full profile for each oil is provided in figures C5.4 and C5.5, together with table C5.4 in appendix C.

From this analysis, it was clear that as determined using endpoint analysis, the cultivation stage provided the over-riding contribution in each of the notable impact areas, however large contributions from the other process stages could be seen in the impact areas of natural land transformation (NLT), freshwater eutrophication (FEu) and climate change (CC). These contributions were all driven by use of power and fuel, with fuel use for the transportation and refining stages leading to relative contributions to NLT of 20.0% and 12.2% for the rapeseed oil and 17.3% and 10.0% for the sunflowerseed oil system.

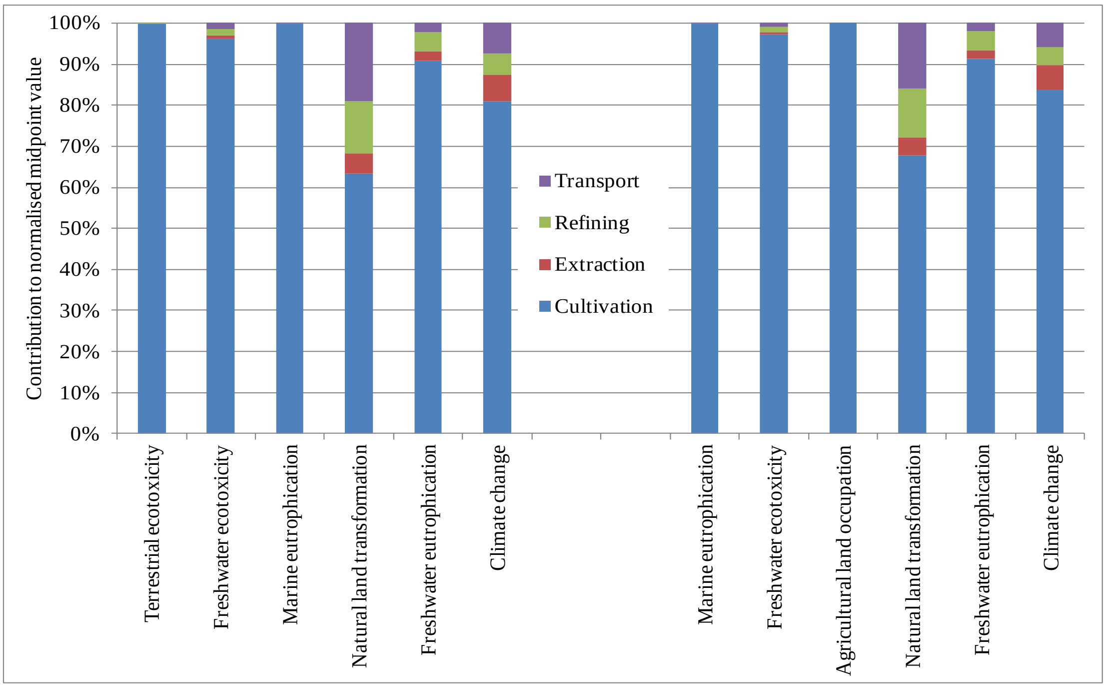

Figure 5.4-5: Relative contributions of process stages to most prominent normalised midpoints

A summary of the relative contributions from each processing stage at both the mid and endpoint level is provided in table 5.4-2. Here it is evident that whether assessed at the mid or endpoint level, the impacts arising from the cultivation stage are dominant.

Table 5.4-2: Percentage contribution to normalised impact values

	Rapeseed oil		Sunflowerseed oil	
	Endpoint	Midpoint	Endpoint	Midpoint
Cultivation	87.8%	94.4%	93.0%	93.6%
Extraction	4.0%	1.1%	2.4%	1.2%
Refining	3.6%	2.1%	2.1%	2.5%
Transport	4.7%	2.4%	2.5%	2.7%

Table 5.4-2 also shows that whereas both mid and endpoint assessment indicate transport as the next largest contributor in both seed oil systems, midpoint analysis places refining as a higher contributor of impacts than extraction, whereas endpoint analysis has the order of these two stages reversed. This will be further highlighted in the next section.

5.4.4. **Contribution to processing**

As cultivation had such an over-riding contribution to the environmental load for both seed oil systems, it was decided to conduct a 'gate-to-gate' (G2G) type analysis to determine where the environmental impacts arose within the processing stages of the system. For this analysis, the starting boundary was receipt at oil mill and the finishing boundary was receipt at food factory. All other parameters were maintained as identical to those from previous analysis.

Figure 5.4-6 shows the breakdown of normalised endpoint impacts per process stage for the G2G analysis, within which it can be seen that for the rapeseed oil system, the transportation required contributed the greatest impacts to the system. This clearly shows the effect of using different transportation methods, since the sunflowerseed system has substantially higher levels of transport included, as detailed in section 5.3.3, however the higher dependence of the rapeseed oil system on road transport rather than sea is driving those impacts up. The effect of different transport scenarios on the model will be fully investigated within section 5.4.6.

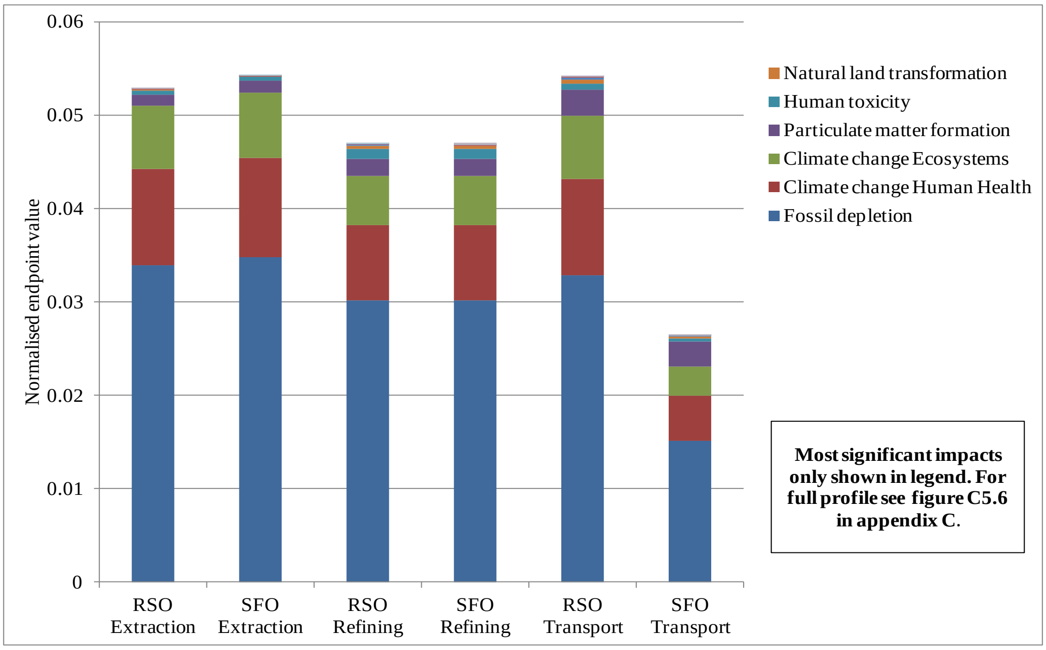

Figure 5.4-6: Process breakdown for normalised endpoint values – gate-to-gate (G2G) analysis

Based on this analysis of endpoints with normalisation using European data, it is evident that the largest impacts arise within the categories of fossil depletion and climate change (eco-systems and human health) and this is consistent within all three processing stages. Within the refining stage, this is predominantly a result of natural gas usage required for steam production which contributes 48.5% of the impact for the rapeseed oil system; however a considerable proportion also comes from diesel usage (20.1%) and electricity use (18.8%).

For the extraction process, the fossil depletion impact comes largely from the use of natural gas for steam production (63.8%), with fossil fuel requirements for electricity production providing most of the remaining impact value. These elements of the refining and extraction processes also drive climate change impacts, both at the human health and ecosystems impact levels.

The G2G analysis was also performed at the midpoint level to ascertain which elements of the unit processes contributed to the most prominent normalised midpoints. The results of this analysis can be seen in figure 5.4-7 where it is evident that within the rapeseed oil system, the impacts from transportation were again higher than either of the individual elements of extraction and refining.

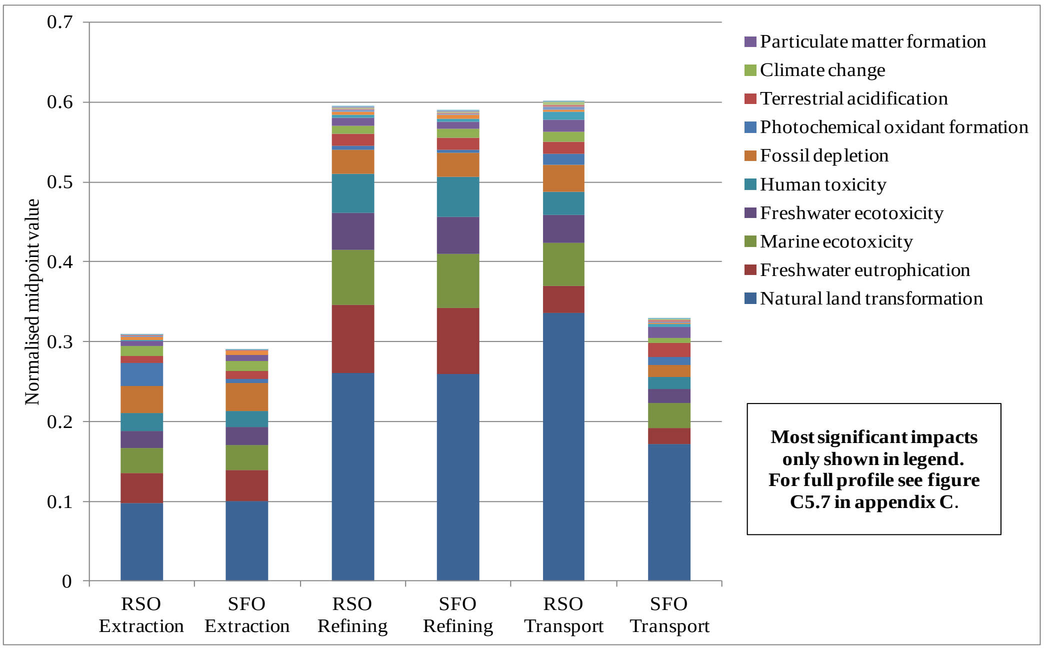

Figure 5.4-7: Process breakdown for normalised midpoint values – gate-to-gate (G2G) analysis

Whilst endpoint analysis placed extraction as the next most contributing process stage for the rapeseed oil system, midpoint analysis identified extraction as least contributing. This stemmed from the different treatment of land use at the mid and endpoint level, where prominent impacts were attributed to the natural land transformation midpoint for the extraction of fossil fuels required. This led to a far greater NLT result in the refining stage than the extraction stage and hence reversed the order of contribution to the seed oil process.

NLT was the most notable midpoint impact for each of the three process stages, when reviewing data normalised with European figures, through the use of fossil fuels within each of the processes, either directly in transport and refining, or to produce electricity and steam as required for extraction and refining. Whilst fossil depletion is an impact in its own right, the land requirement for exploration and extraction of such resources is classified within the NLT midpoint category.

Freshwater eutrophication and marine eco-toxicity were the next two most prominent midpoint categories for the G2G product system. Within these categories, it was again the production of the fuels required within the system that governed the midpoint result, although in this instance it was the disposal methods used during mining and exploration that provided the largest contributions to this category result.

5.4.5. **Sensitivity analysis of alternative methods for treatment of co-products**

Whilst economic allocation was chosen as the method for treatment of co-products throughout the analyses presented in this thesis, an assessment of the level to which the choice of method would affect the results was performed. Initial analysis of the differences obtained when using economic or mass allocation was performed using the Nilsson et al. (2010) cultivation dataset and transportation assumptions accessed from the supporting data of that article.

The findings of this analysis were presented in Hetherington et al. (2011) (reproduced in Appendix A.), within which it was identified that changing from economic to mass allocation caused decreases in the normalised endpoint values of 25% and 30% for the rapeseed cultivation and extraction stages respectively, and increases of 45% and 27% for refining and transport. Changes of similar magnitude took place within the sunflowerseed system changing from economic to mass allocation. Here, the environmental loads attributed to cultivation and extraction both decreased by 30%, with the loads from refining and transport increasing by 45% and 26% respectively.

This analysis was repeated with the amended cultivation data and transport assumptions detailed within this chapter and extended to assess the differences when using system expansion, initially assuming that the co-product meal displaces a mixture of soy and spring barley, as described in chapter 4. The characterised midpoint results for the most prominent impact categories for both seed oil systems are provided in tables 5.4-3 and 5.4-4, from which is evident that changing from economic allocation to system expansion caused increased impact values, whereas changing to mass allocation reduced the impact load.

This result was expected for mass allocation, since the production of more by-products than oil caused the lion's share of the impacts to be allocated to the by-products rather than the oils themselves. To identify the source of the increases through system expansion however and the reason for the notably larger increase in ALO, the changes to the cumulative normalised endpoint value for each processing stage was assessed as shown in figure 5.4-8.

Table 5.4-3: Rapeseed oil system: Characterised midpoint values with different co-product treatment methods

		Refined Rapeseed oil at factory				
		System expansion		Economic allocation	Mass allocation	
		Impact value	% difference from Economic allocation		% difference from Economic allocation	Impact value
CC	kg CO_2 eq	2604	15%	2271	-41%	1340
PMF	kg PM_{10} eq	7.0	32%	5.3	-43%	3.0
TET	kg 1,4-DB eq	161	39%	115	-46%	62
ALO	m^2a	9887	65%	6003	-46%	3230
FD	kg oil eq	525	26%	418	-35%	271

CC= Climate change; PMF = Particulate matter formation; TET = Terrestrial eco-toxicity; ALO = Agricultural land occupation; PMF = Particulate matter formation.

Table 5.4-4: Sunflowerseed oil system: Characterised midpoint values with different co-product treatment methods

		Refined Sunflowerseed oil at factory				
		System expansion		Economic allocation	Mass allocation	
		Impact value	% difference from Economic allocation		% difference from Economic allocation	Impact value
CC	kg CO_2 eq	2998	15%	2597	-44%	1445
PMF	kg PM_{10} eq	5	25%	4	-43%	2
TET	kg 1,4-DB eq	15	35%	11	-48%	6
ALO	m^2a	22080	46%	15158	-48%	7931
FD	kg oil eq	533	26%	424	-39%	257

CC= Climate change; PMF = Particulate matter formation; TET = Terrestrial eco-toxicity; ALO = Agricultural land occupation; PMF = Particulate matter formation.

The results presented indicated that the changes in both seed oil systems were of similar magnitude for all categories except ALO, as explained in the following narrative. The remaining sensitivity analysis data will thus be presented for the rapeseed oil system only.

Analysis showed that when changing to mass allocation, the impact values for the cultivation and extraction stages decreased by 46.2% each, with very modest decreases in contribution from the refining and transport stages. Increased impact values were observed in all process stages when moving to a system expansion approach, when extraction impact value increased by 208 %. This is a result of the way that the soy meal Life Cycle Inventory (LCI) is built to account for co-production.

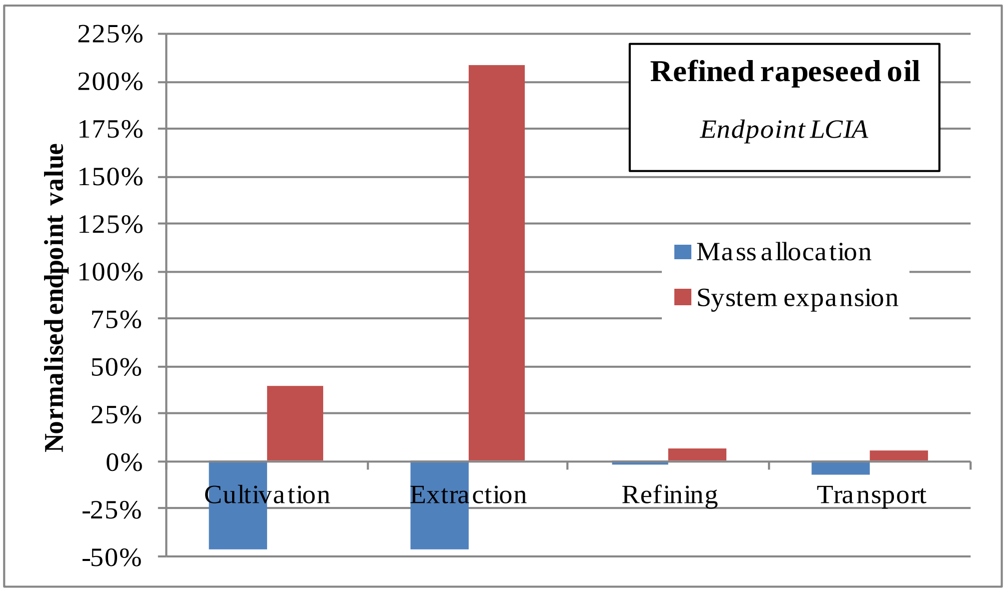

Figure 5.4-8: Relative changes per process stage to cumulative normalised endpoint through changing from economic to mass allocation or system expansion

Soy beans are generally harvested for their meal, with oil produced as a by-product. Within the LCI for soy therefore, the by-product soy oil led to displacement of 0.217 tonne of rapeseed oil per tonne of soy meal, to account for the soy oil that would have been co-produced. This had the effect that within the rapeseed oil system, the 1.5 t of meal co-produced, displaced 1.5 t of soy meal, meaning that 0.326 t of soy oil was not produced, which required an equal figure of rapeseed oil to be produced instead.

This had the net effect that when using system expansion with the rapeseed meal LCI containing soy as the displaced fodder, the normalised endpoint values from the extraction stage increased by 208% compared with the results obtained using economic allocation. This was principally caused by the contribution from ALO derived from rapeseed cultivation for rapeseed oil. Whereas the extraction data using allocation contained minimal ALO impacts, the introduction of an agricultural element through the displacement of soy brought in agricultural scale land use impacts with commensurate dramatic results. The soy displacement effect was reduced for the sunflower system due to rapeseed oil being displaced in the soy LCI rather than sunflowerseed oil which has higher levels of ALO through reduced yields.

An assessment was performed of the results generated through assuming that the co-products displaced alternative sources of fodder, through utilising the following datasets from the LCAFood database:

- Rapeseed meal (containing soy and spring barley in a ratio of 2.38:1)
- Livestock feed (soy)
- Livestock feed (spring barley)

- Livestock feed (wheat)
- Livestock feed (winter barley)

Figure 5.4-9 shows the normalised endpoint values obtained for the refined rapeseed oil LCA when each different approach was used. It was clear that the environmental performance of the rapeseed oil system when using system expansion was highly sensitive to assumptions concerning the animal fodder to be displaced. Results obtained through assuming soy fodder was displaced gave higher impact values than those obtained when using economic allocation. The same results generated assuming both types of barley or wheat were displaced led to lower impact values than those obtained through use of economic allocation. Such inconsistency of results would cause considerable additional uncertainty in the system and it was therefore concluded that choice of economic allocation for ongoing analysis was justified.

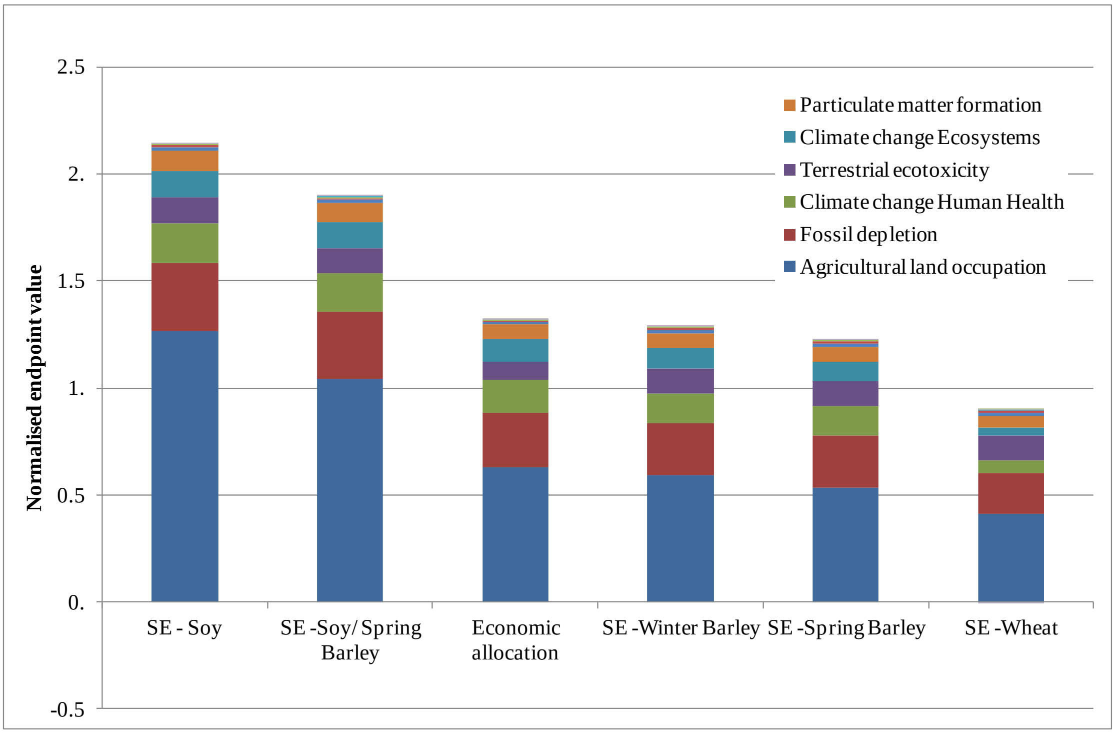

Figure 5.4-9: Comparison of normalised endpoint results for refined rapeseed oil LCIA using alternative displaced animal fodder and economic allocation

5.4.6. **Normalisation variability**

All of the results presented so far were generated using the European normalisation dataset within SimaPro, however as outlined in chapter 4, the ReCiPe (2008) LCIA method gave the option to normalise at the European or World level, with large variations existing between the normalisation parameters for the two regions. Further analysis of both oilseed production models was therefore performed, in line with part (b) of objective 1, to determine the impact that use of different normalisation data would have on the results and estimation of most prominent impact categories.

Figure 5.4-10 shows the results of this analysis for normalisation of endpoints within the rapeseed oil system, with the corresponding graph for the sunflowerseed oil system provided in figure C5.10 in appendix C.

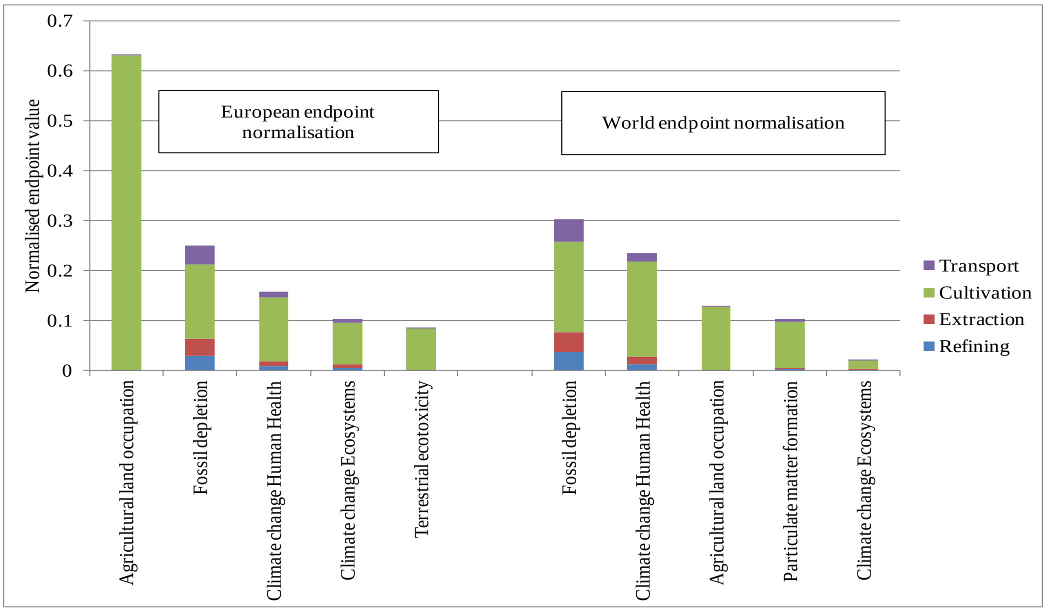

Figure 5.4-10: Impact of changing normalisation data on the five most prominent impact categories for the rapeseed oil system

Whereas normalisation against European values indicates ALO as the most prominent impact category, the same data when normalised with World data places this impact as third, behind fossil depletion and climate change human health. Likewise whilst data normalised with the European dataset identifies TET as fifth most prominent, it isn’t in the top five most prominent endpoint categories when normalised with World values.

The World normalisation value for the eco-systems AoP that ALO and TET reside within is only a fifth of the European normalisation value for eco-systems, causing the endpoint values for these categories to be greatly reduced. As noted by Wegener Sleeswijk et al. (2013) the increased normalisation value for Europe is probably the result of the relatively high percentage of GDP (27%) generated by the European region in comparison to its population. In contrast, since the World normalisation values for the human health AoP are one and a half times those of the European normalisation set, particulate matter formation is the fourth most prominent when viewing the World data, whereas it ranks as sixth within the same data normalised with the European dataset. In this instance, the reduced normalisation value is proposed to stem from the use of cleaner technologies within the EU region (ibid).

The impact of this difference is also clear when viewing the relative contributions from the process stages to the total normalised endpoint result, as shown in table 5.4-4. Here, the rapeseed cultivation data is not indicated as quite as dominating as determined using the European normalisation, with a contribution of 77.7% rather than the previously determined 87.8 %. Similar differences can be seen for the sunflowerseed oil system.

Table 5.4-5: Relative contributions of process stages to total normalised endpoint result with different normalisation parameters

	Rapeseed oil World normalisation	Rapeseed oil European normalisation	Sunflowerseed oil, World normalisation	Sunflowerseed oil, European normalisation
Cultivation	77.7%	87.8%	82.2%	93.0%
Extraction	7.3%	4.0%	6.1%	2.4%
Refining	6.5%	3.6%	5.3%	2.1%
Transport	8.5%	4.7%	6.3%	2.5%

Whilst endpoint normalisation is performed at the damage category (AoP) level, midpoint indicators cannot be aggregated in this way and normalisation of midpoint data is therefore performed using a different reference value for each impact category.

As with the endpoint normalisation, two sets of reference values are available for the ReCiPe impact assessment method, with a choice to normalise at either the European or Global level. As such, an analysis of the effect of using the different reference values was performed when reviewing the midpoint analysis results of the seed oil systems to determine the level to which the midpoint indicators changed when using the two different normalisation sets. The results of this analysis can be seen in figure 5.4-11, within which it is evident that the greatest differences when moving from European to World normalisation data were within the human toxicity, ionising radiation, marine eco-toxicity and freshwater eco-toxicity categories, which produced increased in the normalised impact of 405%, 375%, 252% and 151% respectively.

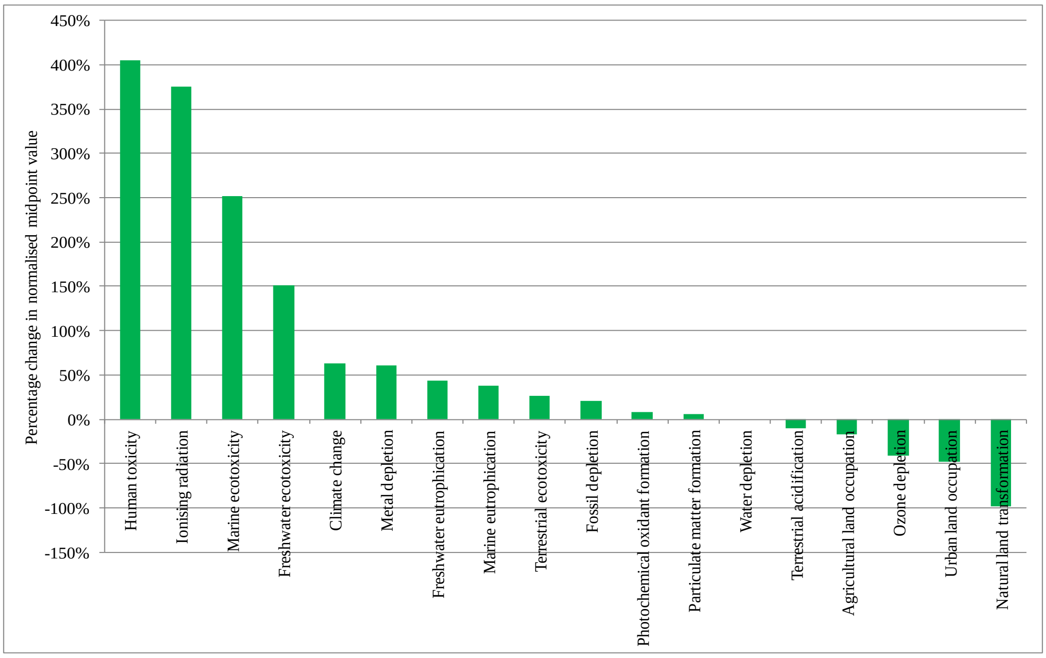

Figure 5.4-11: Percentage change in normalised midpoint value for seed oil LCAs through changing from European to World normalisation datasets

5.4.7. **Transportation**

All data presented so far has been developed using the transport distances described in section 5.3.3, which based cultivation and extraction in Germany for the rapeseed oil system and Spain for the sunflowerseed oil system. In both cases the refining was assumed to take place in The Netherlands before transporting the refined oil via sea and road to a food factory in U.K (Leicester).

As outlined briefly in section 5.4.4, the impacts that arise through differing modes of transport can have differing effects on the result of the assessment. Since the cultivation models used are based on an aggregate pool of seed acquired from locations throughout Europe, the transport distances may be different to those used within the initial assumptions. These effects of increasing the transportation elements were therefore investigated by incrementally increasing both the road and sea transport elements within the sunflowerseed oil system as summarised in table 5.4-5.

Table 5.4-6: Transportation distances used within base case model

Transport	Units	Sunflowerseed	Variance used for sensitivity analysis
Road	km	534	+ 15%
Sea	km	2146	+ 15 to 60%

The variation in contribution to impact results was determined within the five impacts ranked as most prominent through normalisation of endpoint values using European data, excluding the top ranked impact, agricultural land occupation, as this arose exclusively from cultivation. For each of these impact categories (within which the two climate change endpoint categories map to a single midpoint category) the LCA models were analysed to calculate the effect of changing transportation data within the characterised midpoint value as shown in figure 5.4.12.

From this analysis, it is evident that variation in road transportation has a far greater impact on the contribution of the transport element to LCA results than when sea transportation is varied. A 15% increase in road transport yielded a greater impact than increases in sea transport in excess of 60% for all impacts apart from particulate matter formation. To put this into perspective however, the 15% increase in road transport would increase the CFP of the sunflowerseed oil system from 2597 kg CO_2eq to 2616 kg CO_2eq, an increase of 0.7%.

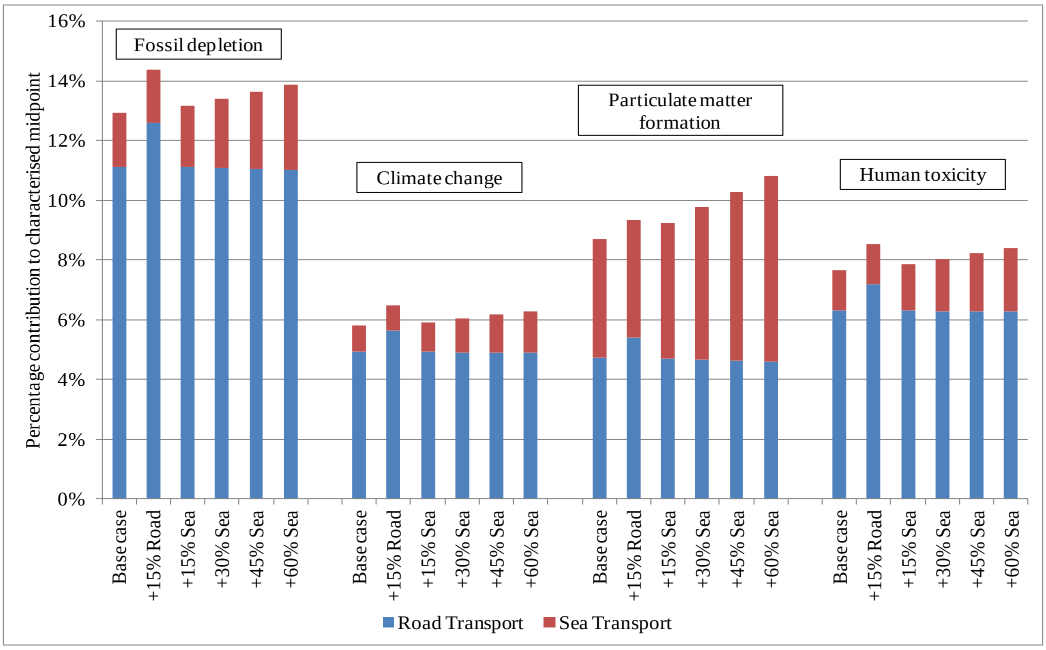

Figure 5.4-12: Variation of characterised midpoints for the sunflowerseed oil system with increased transport distances

5.5. SUMMARY

Attributional LCAs were successfully generated for rape and sunflowerseed oil product systems, from which analyses were performed to fulfil the first of the six research objectives in this thesis. This entailed the quantification of environmental loads and examination of the impact of a range of methodological choices.

Having identified the most appropriate input data and generated a successfully validated LCA model within SimaPro for both seed oil product systems, the CFP and wider environmental loads for production of the refined seed oils were quantified, together with a breakdown of their relative process contributions.

It was clear from analysis at both the endpoint and midpoint levels that the dominant contributor to the environmental burden was the cultivation of the seed. This was the highest contributor within each of the damage categories, to eco-systems, human health and resources, producing between 77% and 88% of the environmental burdens of rapeseed oil, depending on whether the results were normalised against Global or European reference values.

Similar results were identified within the sunflowerseed oil system, where again the cultivation was the primary driver in each area of protection (AoP), with an overall contribution to cumulative normalised endpoint values of 82% to 93% dependant on normalisation set.

With the alternative process for generation of oilseed emulsions examined within this thesis, it was not anticipated that the cultivation stage would be modified in any way, apart from a possible reduction in the need for seed drying, as will be discussed in section 8. Despite the dominant nature of cultivation within the system however, the remaining impacts could all be potentially affected by the new process.

The results have indicated that the main impact contributors to the other processing stages all derive from the consumption of energy in some form, whether from fuel use for transportation, electricity use for extraction and refining or natural gas to produce the steam required for oil extraction. Since the aqueous process for generation of oil emulsions purports to reduce energy consumption, all such reductions will have the potential to generate improvements in those areas. The level to which the perceived benefits come to fruition will be investigated fully within chapter 8.

As part of the wider impact profile, the CFPs for both systems were determined. These were quantified as 2.27 tCO_2eq, for each tonne of refined rapeseed oil delivered to the food factory and 2.60 tCO_2eq for the sunflowerseed oil system. Climate change impacts were not identified as prominent through midpoint normalisation, being placed lower than toxicity, eutrophication and land-use impacts, although endpoint normalisation placed them third and fourth, behind agricultural land occupation and fossil depletion.

As explained by Wegener Sleeswijk et al. (2013) *'large uncertainties remain to exist with respect to toxic substances'* and their prominence within midpoint normalisation should therefore be treated with that uncertainty in mind. As discussed in chapter 4, normalisation of impact values can be subject to an element of uncertainty brought about by incompleteness through lack of emission data, characterisation factors, or both (Heijungs et al., 2007; Kim et al., 2012). Van Hoof et al. (2013) state that ranking based on normalised indicators can vary considerably based on the approach used and support Wegener Sleeswijk et al. (2008) in highlighting toxicity midpoints as potentially having a higher level of uncertainty.

As outlined in chapter 4, the land occupation midpoints are not necessarily impacts, but more a pure measure of land occupied, and the importance of the normalised land occupation midpoint results should therefore be treated with caution. At the endpoint level however the modelling of potentially disappeared species through use and transformation of land makes these more usable as impacts, although the uncertainty introduced through accessing accurate information in this area for normalisation could cause similar uncertainty.

Whilst the issue of uncertainty is independent of where the normalisation references are sourced from, assessment of the differences introduced through using alternative geographical normalisation values (European and Global) found large differences in both magnitude and ranking of the most prominent impacts, highlighting the importance of complete transparency with all methodological aspects of LCA.

The toxicity results varied to the greatest extent when moving from European to World reference values, with Human Toxicity, Marine Eco-Toxicity and Freshwater Eco-Toxicity values changing 405%, 252% and 151% respectively, with only Ionising Radiation producing a change of similar magnitude at 375%. These changes were considerably higher than any of the changes to normalised impact produced within the endpoint comparative analysis, a finding consistent with those of within Lautier et al. (2010) who performed comparisons of both mid and endpoints for European and U.S reference values when developing a set of reference values specific for Canada.

As described by Van Hoof et al. (2013), endpoint normalisation is performed per AoP (Resources, Human Health, Eco-Systems) rather than at the impact level, as is the case for midpoint normalisation. As such there is one normalisation reference value used for each AoP and each of the impacts within that category are normalised against it. Since not all impacts are afforded equal contribution within each damage category, the effect that some of the impacts will have within the AoP reference value are greatly reduced. Endpoints therefore are less sensitive to uncertainties brought about by data incompleteness for impacts that have a low contribution in that endpoint.

They therefore advocate combining the use of normalised endpoint indicators for ranking of significance with characterised midpoint values for reporting results. Given the significance attributed to the toxicity impacts through midpoint normalisation in the seed oil system, it was decided to use this approach for the analyses required in the next chapters, in an effort to limit uncertainty.

Van Hoof et al. (2013), further observed that in normalisation, indicators with a relative high contribution can be considered relevant, but this may differ from what the practitioner perceives to be important. The methodology of normalisation must be taken into account when viewing figures such as those generated here. As outlined in chapter 4, the normalisation score is determined by dividing the characterised result by a reference value, based on the rate of emission of the particular substance of interest created by a single person in Europe (or Globally) during the year 2000.

Comparison of characterised results against a norm established in 2000 could lead to artificially deflated normalisation results for any categories where significant improvements have been made since that year. Considerable efforts have been made to reduce the levels of GHGs since 2000, with many countries imposing GHG reduction targets as part of the Kyoto Protocol established in 1997. Figures published in the 'UK Greenhouse Gas Inventory Report for the period 1990 - 2011' (Webb et al., 2013) indicate an 18% reduction in GHGs in the UK since the year 2000. It is evident therefore that normalised results for climate change may be underestimated due to comparison against a higher historical value.

One of the key methodological choices was with regards to the treatment of co-products within the system. As discussed in chapter 4, the choice had been made to develop attributional LCAs using economic allocation as the preferred method, however to identify the impact that this choice would have on the results, and fulfil objective 3, a sensitivity analysis was performed, using both mass allocation and system expansion.

It was apparent that with the system modelled using the data specified within this chapter, choice of co-product treatment method has a prominent effect on the results. Allocation by mass yielded values below those calculated using economic allocation, and system expansion provided results that were both higher and lower than the economic allocation values, dependant on what source of livestock feed was assumed to be displaced. The uncertainty that would be introduced through using different fodder displacement options for the system expansion model would also be compounded by ambiguity and lack of transparency concerning the boundaries for the data-sets utilised.

It was therefore concluded that the use of system expansion would not be suitable for ongoing analysis and that the economic allocation approach used most commonly within the published oilseed LCAs accessed (Bernesson et al. 2004; Schonfield and Dumelin, 2005; Narayanswamy et al. 2005; Reijnders and Huijbregts, 2008; Stephenson et al. 2008; Nilsson et al. 2010; Stephenson et al. 2010; González-García et al. 2013) was justified for the remainder of this thesis research.

For completeness however, since the magnitude of the changes introduced through moving from economic to mass allocation were smaller than those introduced through the use of system expansion, the use of mass allocation as an alternative method was further explored to assess the impacts for the novel process. This will be detailed in chapter 8 where the comparison between the novel and existing process takes place.

5.6. CONCLUSIONS

This chapter presented the findings of research objective 1, using LCA to establish the environmental loads for the production of refined rapeseed and sunflowerseed oils, identifying the relative contributions from each of the processing stages and using this data to examine the impact of allocation and normalisation choices.

As detailed within this chapter, having validated the cradle to gate LCA models developed for both seed oils against published results from literature, the data utilised was augmented and improved prior to performing a range of impact assessment analyses. The environmental burdens of the seed oil systems were fully identified using both endpoint and midpoint category indicators within which the CFP of each tonne of refined rapeseed oil delivered to the food factory was determined as 2.27 tCO_2eq and the corresponding CFP for the sunflowerseed oil system was 2.60 tCO_2eq per tonne of refined oil delivered to food processor.

Sensitivity checks were performed for the key methodological choices of co-product treatment and normalisation, together with the impact of varying transportation requirements. Based on these findings, the parameters used within the models developed and analysed in this chapter will be retained for all other LCA work within this thesis, such that unless otherwise stated, cultivation data will be based on the use of the aggregate data sets described here, allocation will be by economic value, and normalisation will use European reference values.

Having successfully developed the LCA models to establish the environmental loads for the production of both seed oils, together with the relative contributions from each of the processing stages, this model will be utilised as a raw material within the case study emulsion product, mayonnaise. The modelling and analysis work for this will be detailed within the next chapter.

CHAPTER 6. LCA OF MAYONNAISE

In order to fulfil the research objectives set out in chapter 1, a case study food product was chosen to guarantee functional equivalence of the comparison. As stated in chapter 2, whilst mayonnaise is potentially the simplest of all food emulsion products, Depree and Savage (2001) state that it is probably one of the most widely used condiments in the world today and was therefore an ideal product for this purpose.

This chapter outlines the use of Life Cycle Assessment (LCA) to identify the environmental loads of the current processing route for standard mayonnaise production using rape and sunflowerseed oils to determine the full range of environmental impacts, in line with research objective 2. In addition, results are presented for the investigation into the appropriateness of using the single issue LCA variant, Carbon Footprinting (CFP) as an environmental performance measure for this system.

This data will then be used in chapter 8, both to determine the areas where the novel process can afford savings and to provide a benchmark for comparison. Specifically, the analysis will provide the following outputs:

a. Identification of the environmental profile and carbon footprint of commercially produced mayonnaise using rape and sunflowerseed oils
b. Determination of the contribution from each process stage to the impacts of the mayonnaise process using both types of seed oil
c. Provision of clarity on whether the focus on climate change impacts through carbon footprint reduction measures would lead to burden shifting within the mayonnaise production system.

The composition of mayonnaise varies, but has set minimums for oil content with the US Food and Drug Administration (FDA) regulation (21CFR169.140) stipulating a minimum of 65%, whilst the European Federation of Condiment Sauce Industries (FIC) recommends a minimum of 70%. In practice, commercially produced mayonnaise has a fat content of 70 – 80% (Garcia et al., 2009). Despite its high fat content, it is an oil-in-water emulsion produced using different types of oil, dependant on brand and geographical location. As outlined within Ma and Boye (2013), Martin et al. (2000) cite that the types of oil commonly used in the formulation of dressings and mayonnaise include soybean, canola, and sunflower oil, and sometimes cottonseed and olive oil.

Within Europe, rapeseed oil and sunflowerseed oil are the most prevalent for mayonnaise production and an LCA model was therefore developed to investigate the cradle-to-gate production system of conventional mayonnaise, utilising the LCA models detailed in chapter 5 to represent the rape and sunflowerseed oils used.

6.1. **RELEVANT LITERATURE**

There are many journal articles and patents covering different production techniques for mayonnaise including Dartey et al. (1990), Takashi and Hiroko (1999), Garcia et al. (2009), Bengoechea et al. (2009) and Kerkhofs et al. (2011). There are also several publications covering the physical and flavour characteristics of mayonnaise such as Depree and Savage (2001), Guilmineau and Kolozik (2006), Marayama et al. (2006) and a plethora of papers detailing methods for reducing the fat content of the world's favourite food emulsion. Whilst these are all useful for understanding the characteristics and processing of the material, and could therefore have a bearing on the scope of the LCA, they were unable to yield any information for inventory creation or validation purposes.

A search of the bibliographic databases 'SCOPUS', 'Web of Science' and 'Compendex' for publications of any kind relating to the environmental impacts of mayonnaise yielded only two results; Adenugbaac et al. (2008) who researched the levels of polychlorinated biphenyls (PCBs) in several food products including mayonnaise and Katami et al. (2004) who investigated the formation of dioxins from the incineration of a variety of food products, including mayonnaise, that could be found in domestic waste. Searching for any material combining the terms 'mayonnaise' and 'LCA' yielded no results in any search portal, and no information could be found in any of the LCI databases.

As noted in chapter 3, the strong focus on carbon reduction initiatives over the past few years has led to the popularity of the single-issue LCA variant carbon footprinting (CFP) soaring. Within the UK, Food and Drink Federation (FDF) members are committed to an industry-wide absolute target to reduce CO_2 emissions by 35% by 2020 against a 1990 baseline measured within their voluntary Climate Change Agreement with the Department for Energy and Climate Change (DECC) (www.fdf.org.uk, 2013). A search was therefore also performed to identify whether any CFP data could be accessed for mayonnaise, since this may provide a route to access data. Unfortunately the only reference to CFP for mayonnaise was found on a web page for fast moving consumer goods (FMCG) manufacturer and mayonnaise producer Unilever, who compared their 'light mayonnaise' with their standard version and stated that the former has *'a very positive impact on sustainability. Reducing the fat content from 75% to 25% decreases the carbon footprint by roughly 40%.'* (www.unilever.com, 2013). However no actual figures were provided for the CFPs themselves.

With such a lack of information available it is clear that in addition to the modelling of the system using LCA being an essential stage within this research project, publication via this thesis of the data generated, will place information into the public domain that was hitherto unavailable.

6.2. SYSTEM DEFINITION

The product system to be analysed within this case study is the production of commercially available mayonnaise using conventional techniques. The functional unit (FU) used for the analysis was '1 tonne of rapeseed / sunflowerseed oil mayonnaise produced in UK, packaged in 600g jars, palletised and ready for distribution'. As a cradle to gate study, the starting boundary was the extraction of raw materials, which translated to cultivation of crop and rearing of animals for the agricultural products involved. The finishing boundary was the exit from the mayonnaise packaging facility, thereby excluding use and disposal stages of the life cycle. The system flow digram is shown in figure 6.2-1.

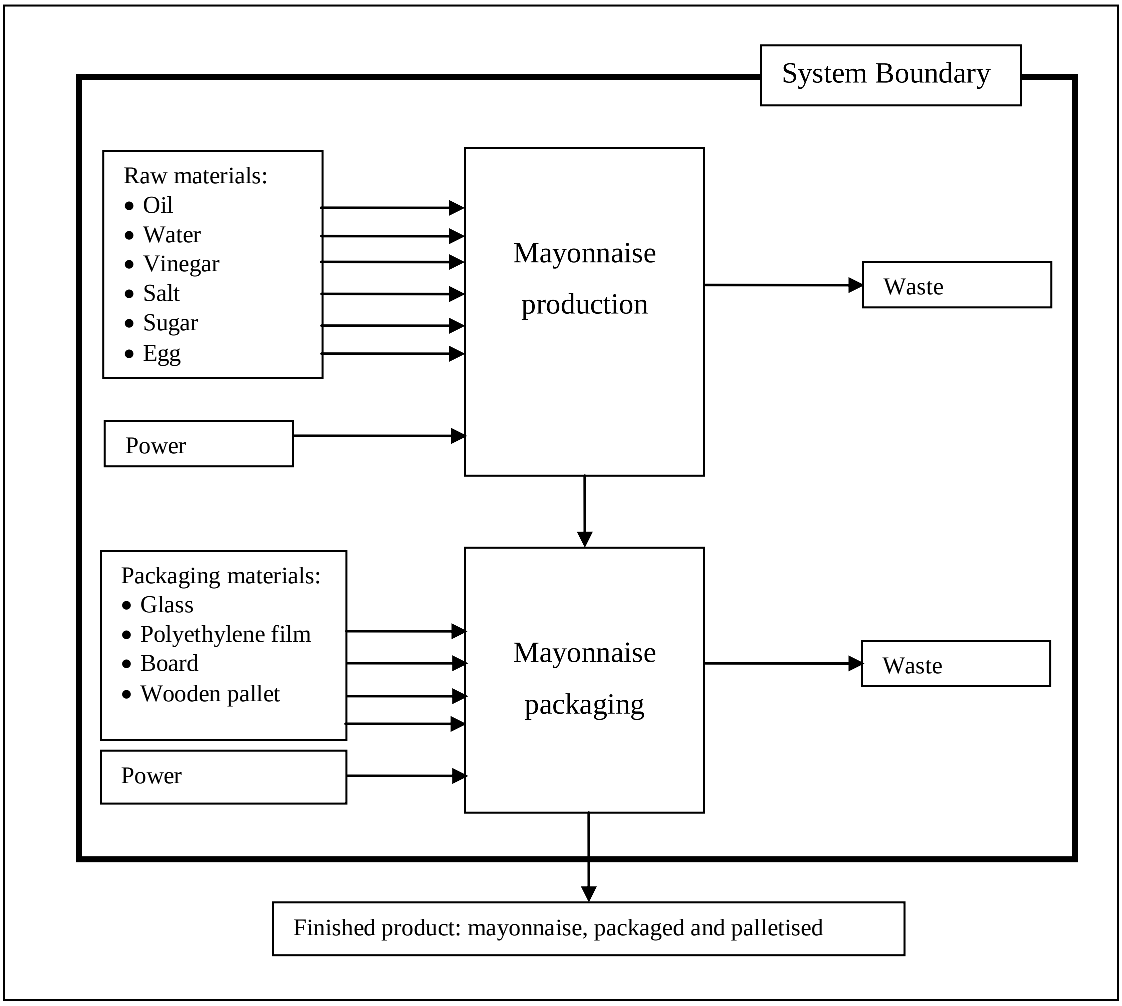

Figure 6.2-1: System flow diagram for mayonnaise LCA

Attributional LCA models were constructed within SimaPro 7.3.2.for both mayonnaise systems incorporating rapeseed and sunflowerseed oil, using the composition as shown in table 6.2-1.

Table 6.2-1: Mayonnaise composition used for LCA, Source: adapted from Meeuse et al. (2000)

Ingredient	Fraction	Function
Oil	80%	Emulsion formation
Egg yolk	8%	Increase stability
Water	7%	Emulsion formation
Vinegar	3%	Taste, preservation, increase stability
Salt	1%	Taste, increase stability
Sugar	1%	Taste

As outlined within chapter 2, the basic steps for mayonnaise preparation, involve adding the surfactant (egg) to the water and mixing the solution with an equal volume of oil to form a crude emulsion. The emulsion is then passed through a colloid mill or homogeniser, with more oil being incorporated as required to generate the final product, which is passed on to the bottling and further packaging stages. Packaging generally involves the application of labels, fitting of caps and aggregation of the bottles into packs, using cardboard and polythene film, followed by palletisation using a pallet and shrink-wrap film. Details of packaging assumptions used can be found in appendix B.

For the purposes of this model, the manufacture and packaging of the mayonnaise was assumed to take place at the same manufacturing facility. The data for energy and water consumption at the mayonnaise facility was unpublished data which was sourced from industry as data ‘per tonne of “packed finished product” ready for distribution’. It was not possible to obtain data for each individual element of the facility e.g. emulsification stage, packaging process, but the aggregate data supplied was sufficient for the purposes of this study. The power consumption and water usage data that was provided represented an average from the production units of the mayonnaise company in a range of different geographical locations. For the water used, the data did not differentiate between water used as an ingredient and that used for processing stages such as cooling and cleaning.

As previously mentioned, the models outlined in chapter 5 were used for the seed oil portion of the ingredients, with life cycle inventory (LCI) data for the remaining constituents, packaging materials and energy production sourced from peer reviewed literature and proprietary databases available within SimaPro. The sources of data for the required LCI datasets are outlined within table 6.2-2.

Table 6.2-2: Details of data inputs for mayonnaise process model.

Process	LCI data source	Module title within SimaPro	Reference
Seed oil			
Rapeseed cultivation	EcoInvent unit processes, to develop aggregate data set	25% of each of the following: Rape seed conventional, Saxony-Anhalt, at farm/DE Rape seed conventional, Barrois, at farm/FR Rape seed extensive, at farm/CH Rape seed IP, at farm/CH	Nemecek et al. (2007) Nemecek et al. (2007) Nemecek et al. (2007) Nemecek et al. (2007)
Sunflowerseed cultivation	EcoInvent unit processes, to develop aggregate data set	50% of each of the following: Sunflower conventional, Castilla-y-Leon, at farm/ES Sunflower IP, at farm/CH	Nemecek et al. (2007) Nemecek et al. (2007)
Extraction	Industry data amended to UK energy mix*		Unpublished industry data
Refining	Industry data amended to UK energy mix*		Unpublished industry data
Sugar			
Cultivation	EcoInvent unit process	Sugar beets IP, at farm/CH	Jungbluth et al. (2007)
Processing	EcoInvent unit process	Sugar, from sugar beet, at sugar refinery /CH	Jungbluth et al. (2007)
Egg production	LCAFood database	Egg	http://www.lcafood.dk (2013)
Salt manufacture	EcoInvent unit process	Sodium chloride, powder, at plant/ RER	Althaus et al. (2007)
Vinegar manufacture	Based on EcoInvent unit process for acetic acid	6% :Acetic acid, 98% in H20, at plant/RER U 94%: Water, deionised, at plant/CH	Althaus et al. (2007) Althaus et al. (2007)
Water (for formulation and general site use)	EcoInvent unit process	Tap water, at user/RER	Althaus et al. (2007)
Packaging glass manufacture	EcoInvent unit process(including 60% recycled material)	Packaging glass, white, at plant/RER	Hischier (2007)
Packaging film manufacture	EcoInvent unit process	Packaging film, LDPE, at plant/RER	Hischier (2007)
Packaging board manufacture	EcoInvent unit process	Packaging, corrugated board, mixed fibre, single wall, at plant/ RER	Hischier (2007)
Euro - Flat Pallet	EcoInvent unit process	EUR-flat pallet / RER	Kellenberger et al. (2007)
Power generation	EcoInvent unit process	Natural Gas, burned in mini CHP plant /CH	Heck et al. (2007)
Transport- Road	EcoInvent unit process	Transport, lorry 16-32t, Euro5/RER	Spielman et al. (2007)
Transport- Sea	EcoInvent unit process	Transport, transoceanic freight ship/ OCE	Spielman et al. (2007)

* - UK energy mix dataset developed using Digest of UK Energy Statistics (DUKES) data combined with EcoInvent unit processes.

For life cycle impact assessment (LCIA), ReCiPe(2008) was used, with analysis performed using the hierarchist version at both midpoint and endpoint levels. Since the ReCiPe characterisation model for GWP is identical with that from IPCC (2007), ReCiPe GWP data was extracted to generate CFPs for both systems. Where normalisation was performed, European reference values for the year 2000 were used, based on the work of Wegener Sleeswijk et al. (2008).

6.3. RESULTS AND DISCUSSION

Having created the LCA models using the methodology outlined, life cycle impact assessment (LCIA) was used to determine the following information required for research objective 2:

1. The environmental loads of mayonnaise produced using rape and sunflowerseed oils
2. The elements of the process that were the primary contributors to the prominent environmental loads

This information could then be used to identify environmental hot-spots within the system prior to the comparison with the LCA results of the mayonnaise-like emulsion produced using oil-body material within chapter 8. Overview results data is presented within this chapter, with further detailed information available within appendix D.

6.3.1. Environmental loads for the mayonnaise systems.

Normalised endpoint results for mayonnaise produced with both types of oil are shown in figure 6.3-1, within which the most prominent impacts are shown in the legend. From this it can be seen that the largest normalised endpoints arise within the Damage to Ecosystems area of protection (AoP), with Damage to Resources having the next largest cumulative impact for both mayonnaise variants. As with the seed oil LCA results presented in chapter 5, the impacts for the sunflowerseed mayonnaise system are larger than those of the rapeseed system, with the value calculated for agricultural land occupation (ALO) causing the sunflower mayonnaise (SFM) Ecosystems AoP to have impacts almost twice those of the rapeseed variant.

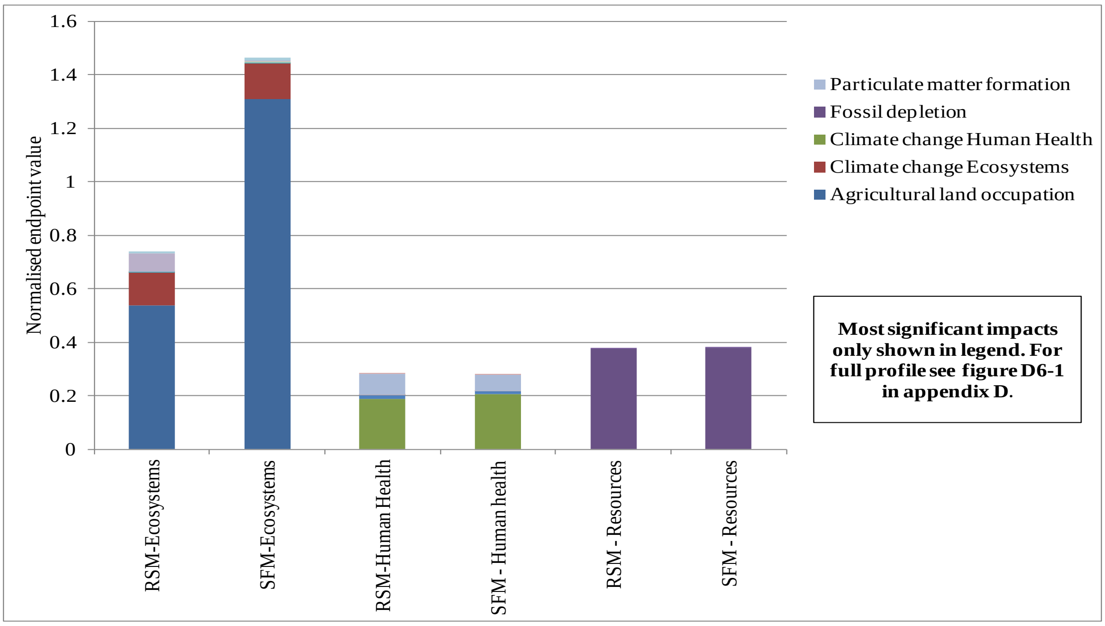

Figure 6.3-1: Normalised endpoint results for mayonnaise systems. RSM = Rapeseed oil mayonnaise; SFM = Sunflowerseed oil mayonnaise

For the sunflowerseed mayonnaise, 97.6% of the ALO impacts came from the seed oil, with a further 1% derived from egg production. With the reduced magnitude of ALO impacts within the rapeseed mayonnaise, the same impact from the egg production provided a 2.5% contribution, with 94.1% coming from the rapeseed oil. With such a large contribution from cultivation of the seeds, the larger acreage required for producing sunflowerseed compared with that of rapeseed, as highlighted in Shonfield and Dumelin (2005) and supported by Iriarte et al. (2010) was clearly shown within the results here.

Within the resources AoP, the seed oil system provided 52.2% of the fossil depletion (FD) impact for the sunflowerseed mayonnaise, with the contribution from glass production providing the next largest, at 24.2%. The figures for the rapeseed system were similar at 51.7% for the oil and 24.4% for the glass. Further analysis of the process contributions will be presented in section 6.3.2.

The characterised midpoint results for both mayonnaise systems are shown in table 6.3-1, from which it can be seen that the CFP for rapeseed oil mayonnaise is 2.69 tonnes CO_2eq, with the sunflower variant having a larger CFP of 2.96 t CO_2eq. This is consistent with the higher CFP of sunflowerseed oil compared with rapeseed oil, as a result of lower sunflower yields as discussed in section 5.5.2.

Table 6.3-1: Characterised impacts of rape and sunflowerseed mayonnaise

Impact	Units	Rapeseed Mayonnaise	Sunflowerseed Mayonnaise
Climate change	kg CO_2 eq	2694	2955
Ozone depletion	kg CFC-11 eq	0.0003	0.0002
Human toxicity	kg 1,4-DB eq	452	347
Photochemical oxidant formation	kg NMVOC	9.6	11.4
Particulate matter formation	kg PM_{10} eq	6.0	4.7
Ionising radiation	kg U_{235} eq	245	279
Terrestrial acidification	kg SO_2 eq	29.5	16.4
Freshwater eutrophication	kg P eq	0.7	0.7
Marine eutrophication	kg N eq	23.1	70.6
Terrestrial eco-toxicity	kg 1,4-DB eq	92.5	8.9
Freshwater eco-toxicity	kg 1,4-DB eq	27.6	34.9
Marine eco-toxicity	kg 1,4-DB eq	8.9	8.5
Agricultural land occupation	m^2a	5210	12539
Urban land occupation	m^2a	51.9	83.5
Natural land transformation	m^2	0.5	0.5
Water depletion	m^3	17.5	16.5
Metal depletion	kg Fe eq	96.7	96.8
Fossil depletion	kg oil eq	624	630

The normalised midpoint impacts for both types of mayonnaise are shown in figure 6.3-2. In both cases the most prominent midpoint impact was the same as for the seed oil system alone, with terrestrial eco-toxicity (TET) having the largest normalised midpoint value for the rapeseed mayonnaise and marine eutrophication (ME) identified as most prominent for the sunflowerseed variant.

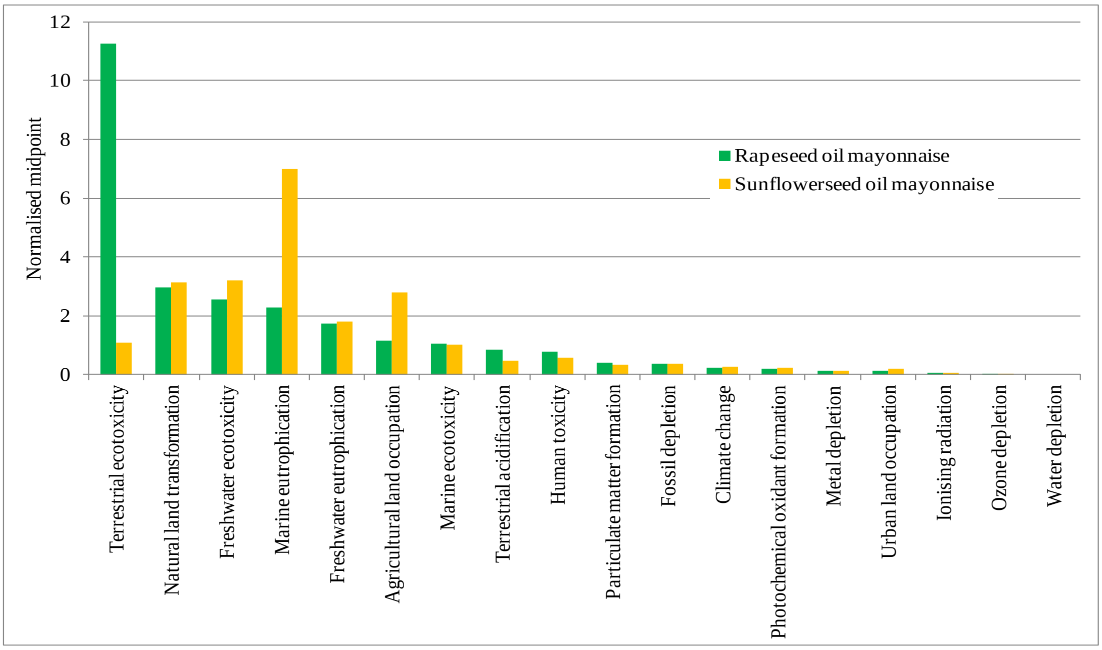

Figure 6.3-2: Normalised midpoint values for rape and sunflowerseed oil mayonnaise systems

As with the seed oil LCIA results, the impacts identified as most prominent through midpoint normalisation are different to those indicated through endpoint normalisation, with toxicity and eutrophication impacts being determined as having a high level of importance. Seed oil inputs do not completely dominate the results however, as evidenced by the increased significance of natural land transformation (NLT) compared with the seed oil systems alone. The increased ranking from fourth in both seed oil systems to second in the rape mayonnaise system and third in the sunflower mayonnaise system is derived from the extraction of fossil fuels required for glass production.

In addition to having higher results within the marine eutrophication midpoints, the sunflowerseed oil mayonnaise had much larger results for ALO as a result of the lower yields. Within ALO, cultivation of the rapeseed provided 91.5% of the impact results and sunflowerseed provided 96.8%.

6.3.2. Carbon footprint data for both mayonnaise variants

Normalised climate change impacts did not rank as most prominent when the LCA model was assessed using either midpoint or endpoint techniques. As discussed in the previous section, when reviewing normalised midpoint data shown in figure 6.3.2, it was twelfth out of the eighteen midpoints. Within the endpoints results however, the impact of climate change on human health and the impact of climate change on eco-systems featured as third and fourth among the seventeen indicators, indicating a higher level of significance than that attributed through midpoint analysis.

This is consistent with the findings discussed in chapter 5, where it was acknowledged that the higher degrees of uncertainty associated with the toxicity midpoints, the fact that land occupation 'impacts' were not true impacts and the use of a normalisation reference year of 2000 would all have the potential to artificially deflate the climate change normalised result in comparison with the other categories.

As indicated in section 6.1, no publicly available data could be found for the environmental credentials of mayonnaise and despite the popularity of the single issue LCA variant CFP for consumer goods there was no information for this either. In light of the UK, Food and Drink Federation (FDF) members' commitment to an industry-wide absolute CO_2 reduction target of 35% against a 1990 baseline by 2020, it is therefore valuable to extract the CFP values from the LCA, together with the constituent contributors, to be used to identify whether the information obtained from the CFP alone would enable process improvements to be targeted correctly, or potentially cause burden shifting. As the GWP characterisation method within ReCiPe (2008) uses the IPCC (2007) equivalence factors (Goedkoop et al., 2013) and as the scope of the assessment conforms to PAS2050:2011, the climate change impact category results therefore provided the CFP for the two systems.

From the characterised data presented in table 6.3.1 it was evident that the CFP of one tonne of packaged, palletised mayonnaise (FU) produced with sunflowerseed oil was 2.96 t CO_2eq, and the CFP of the rape seed oil variant was slightly lower, at 2.69t CO_2eq per FU.

Analysing the systems further, the breakdown of the CFPs can be seen in figure 6.3.3. Here it is evident that the largest single contributor to CFP was the seed oil for both types of mayonnaise, contributing 62.2% of the impacts for the sunflowerseed oil mayonnaise and 58.4% of the impacts for the rapeseed oil mayonnaise.

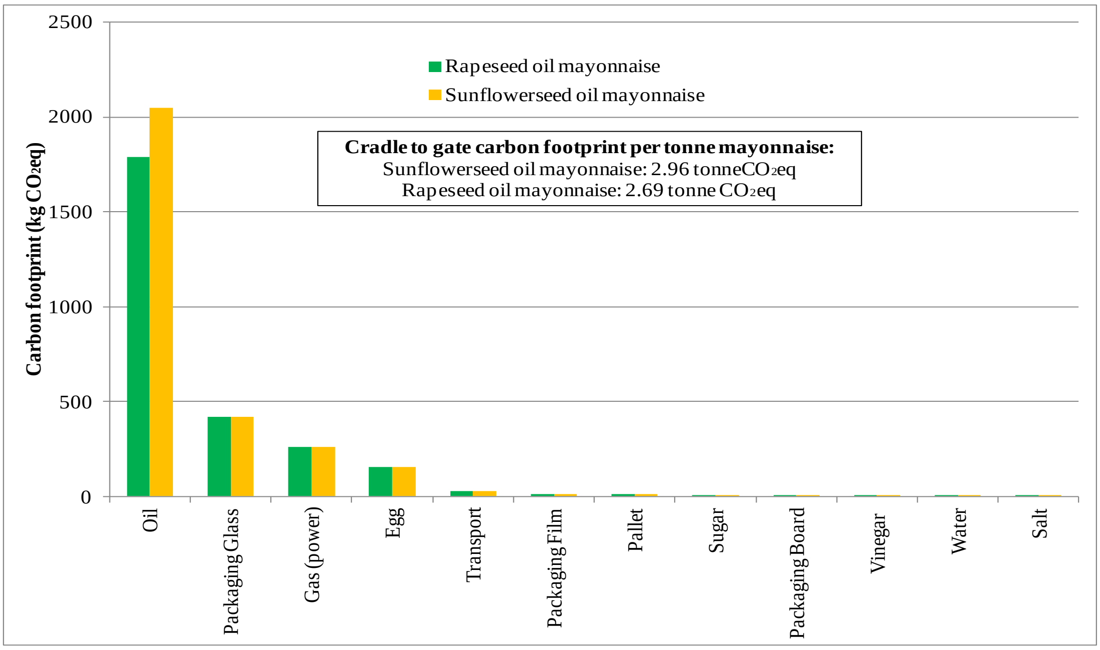

Figure 6.3-3: Relative contribution of process elements to CFP of mayonnaise

The second largest impact category for both types of mayonnaise was packaging glass, which yielded a CFP of 0.4 t CO_2eq per FU, representing a contribution of 14.2% of the CFP for the sunflowerseed oil mayonnaise and 15.5% for the rapeseed oil mayonnaise. It should be noted that the data for packaging glass takes into account recycling of the glass used as the dataset utilised incorporated 60% recycled glass as part of the glass production process. The third largest contributor to CFP was that of power consumption at the manufacturing plant, contributing 0.3 tCO_2eq per tonne of packaged and palletised mayonnaise, representing an 8.8% contribution to the sunflowerseed oil system and a 9.7% contribution to that of rapeseed oil. As outlined in section 6.2, the power consumption utilised here included not only the processing power for emulsification, but all energy inputs associated with production of the FU and was an average figure based on production units in different locations.

Drilling deeper into the results for the largest category, that of seed oil data; it was clear that in both cases, the cultivation of the seed provided the greatest contribution to the seed oil CFP, with 87.2% of the sunflowerseed oil CFP coming from the cultivation stage and 82.7% of the rapeseed oil CFP coming from that stage. Shonfield and Dumelin (2005), comment that *'Sunflower oil tends to have high environmental impacts because of the relatively low yields per hectare compared to other crops'* and it is this higher impact that causes the CFP of sunflowerseed oil derived mayonnaise to be higher than that of its rapeseed oil counterpart.

6.3.3. **Relative impacts of process stages**

Having quantified the environmental impacts for both types of mayonnaise at the mid and endpoint level, it was important to identify the contributions that each element of the process made to the calculated impacts, to build understanding of where process improvements could best be targeted to yield environmental improvements. Since the categories identified as most prominent through midpoint normalisation were those midpoints that attract the highest levels of uncertainly through incomplete emissions and characterisation data, the approach advocated by Van Hoof et al. (2013) of reviewing the characterised midpoints for the impact categories identified as prominent through endpoint normalisation was used for this analysis.

Figure 6.3.3 indicates the percentage contributions to each of the characterised midpoint impacts within the rapeseed mayonnaise system, showing impact categories with the highest significance (through endpoint normalisation) on the left and lowest on the right.

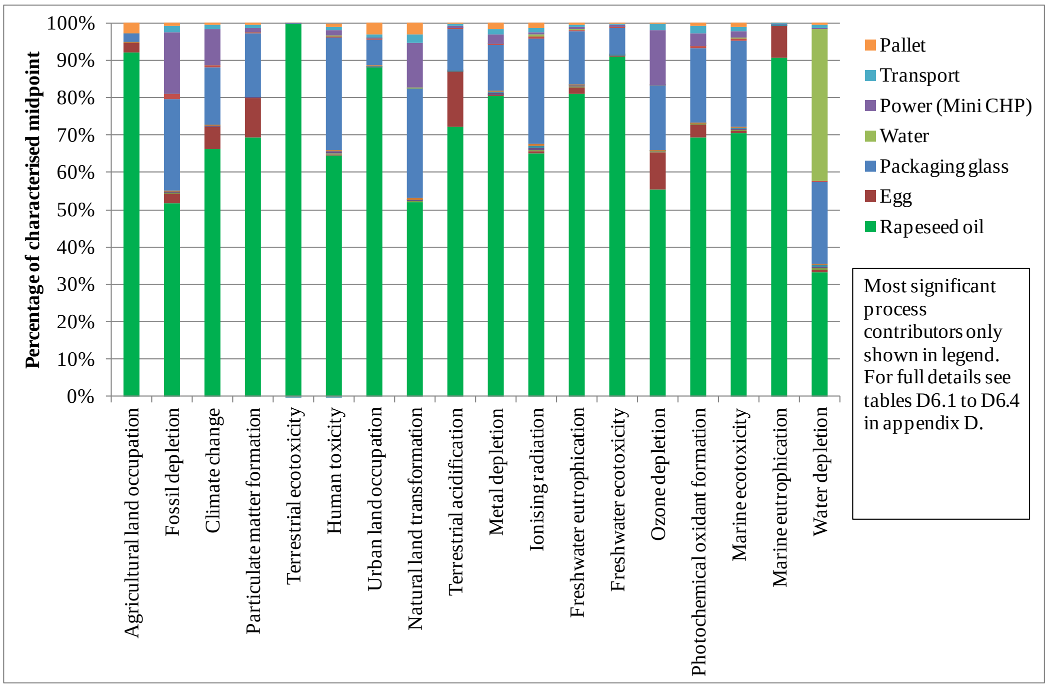

Figure 6.3-4: Percentage contributions to characterised midpoint data: Rapeseed mayonnaise

Since the only variable within the two systems was the seed oil, all other contributors had equal amounts of characterised impacts irrespective of which seed oil was used for the mayonnaise, although the percentage contributions to the different systems varied, due to the total impacts of the system being different. Full data for this can be found in tables D6.1 to D6.4 in appendix D.

Whereas the most prominent midpoint impact category for rapeseed oil based mayonnaise was terrestrial eco-toxicity, endpoint normalisation relegated this to fifth, with agricultural land occupation being identified as the most burdensome category. Within this the result was dominated by the impacts brought in by the seed oil system which provided 92.2% of the ALO impacts. This was almost exclusively from the cultivation of seed used to produce the oil. Apart from TET, where the impacts derived from seed cultivation provided 99.8% of the burdens, ALO was the midpoint most dominated by seed oil impacts.

The seed oil impacts did not dominate all of the most prominent impact categories however, with fossil depletion (FD) impacts, which were calculated as second most prominent, only deriving 51.8% of their impacts from the seed oil system. Within this category, substantial contributions also arose from packaging glass (24.4%) and the use of power within the mayonnaise manufacturing plant (16.3%).

Whilst midpoint normalisation had ranked climate change as twelfth out of the seventeen normalised midpoints, endpoint normalisation placed the two climate change (CC) endpoints as third and fourth. Within the single characterised climate change midpoint, similar contributions to the fossil depletion category were found, with the seed oil having the greatest level of burdens at 66.4% of the midpoint value, but packaging glass and power usage at the mayonnaise plant also providing considerable contributions of 15.5% and 9.7% respectively.

The fourth most burdensome impact area through endpoint normalisation was particulate matter formation (PMF). Whilst the largest contributions in this category again stemmed from the seed oil (69.5%) and packaging glass (16.9%) the third highest contributor in this category was egg use (10.5%). This result, which is perhaps surprising, given the modest use of egg as a raw material, was driven by the liberation of ammonia and nitrous oxide during poultry farming.

The percentage contributions to each of the characterised midpoint impacts within the sunflowerseed mayonnaise system are shown in figure 6.3.5, which again depicts the impact categories in order of decreasing significance when viewing data normalised at the endpoint level, from left to right.

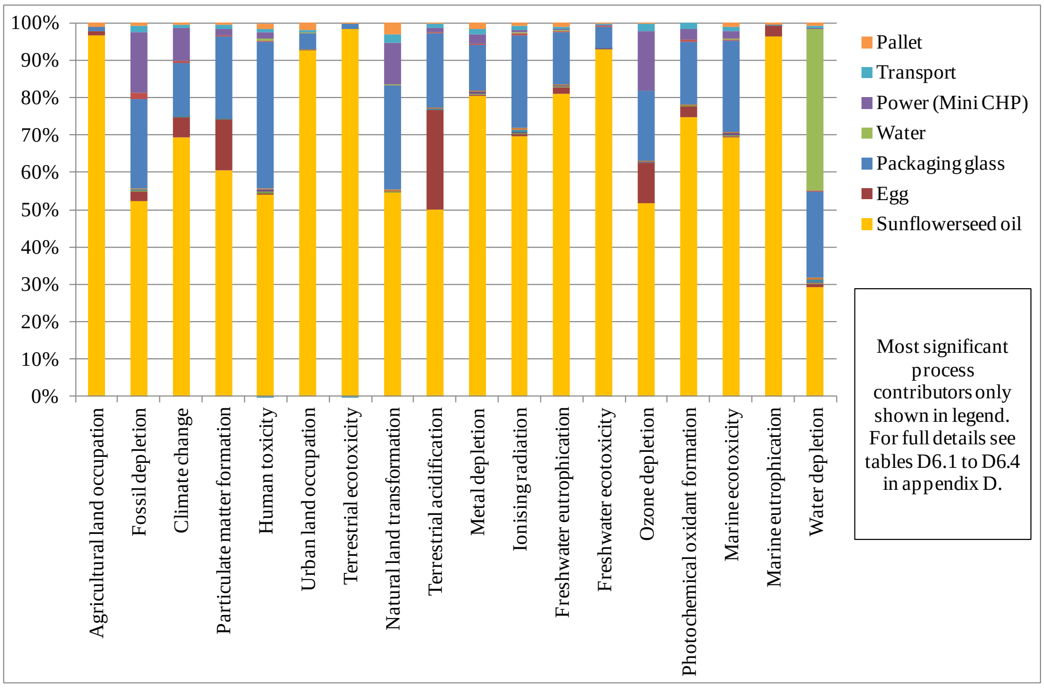

Figure 6.3-5: Percentage contributions to characterised midpoint data: Sunflowerseed mayonnaise.

As with the rapeseed mayonnaise system, the four most prominent impact categories were ALO, FD, CC and PMF, which largely had similar process contributions from each of the processing stages. In this case however, the fifth largest impact category was human toxicity (HT) within which the largest contributions arose from seed oil, at 54.3% of the impact and packaging glass, which contributed a prominent 39.4% of the impact value. These burdens derived from the chemical emissions such as arsenic, selenium and lead during glass manufacture, in addition to the waste streams from lignite use within the fuel mix.

To more clearly determine the level to which each process element contributed to the overall environmental impacts of the mayonnaise system, the normalised impacts were aggregated for each process input. This can be seen in figure 6.3.6, from which the dominating contributions of the seed oils are clearly seen.

Note that both sunflower and rapeseed oil are depicted on this graph, to illustrate the magnitude of normalised impacts in comparison with each other. This is possible since as previously noted; the only variable within the system is the seed oil, with all other components remaining unchanged.

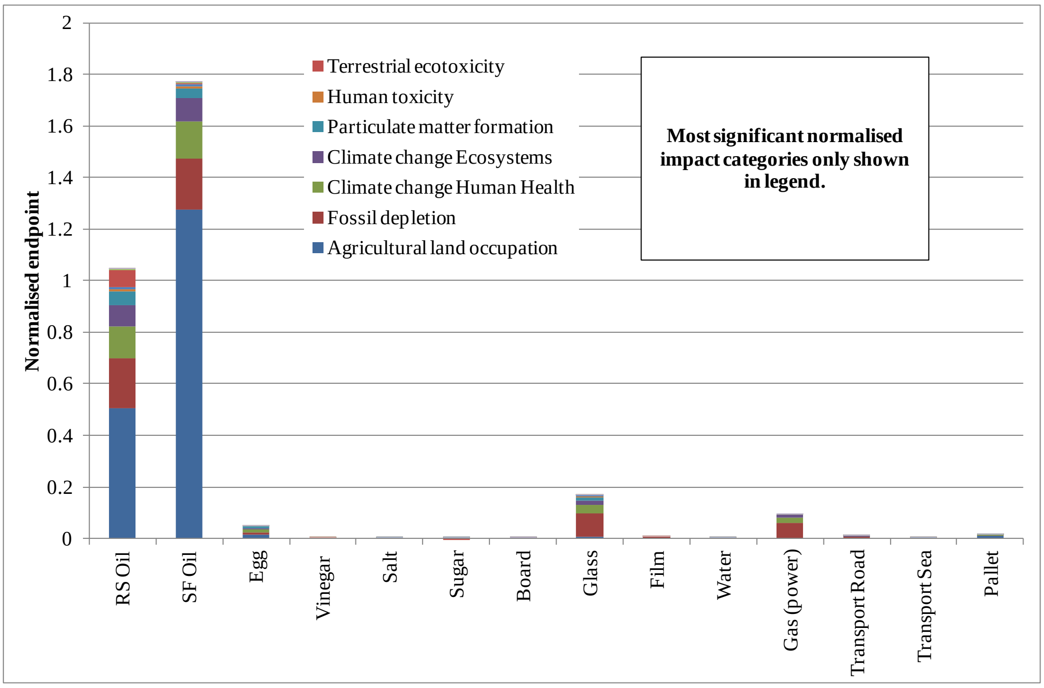

Figure 6.3-6: Aggregation of normalised endpoints for mayonnaise system

6.4. SUMMARY

Attributional LCA models were successfully created for both rape and sunflowerseed oil based mayonnaise, such that the outcome of objective 2 could be obtained. This entailed both the determination of environmental loads and the examination of the impact of methodological choices.

From the analysis presented here, it was clear that for the system representing '1 tonne of rapeseed / sunflowerseed oil mayonnaise produced in UK, packaged in 600g jars, palletised and ready for distribution', the contribution from the seed oil dominated the environmental impacts attributed to both types of mayonnaise. In both cases, the most prominent impacts identified through both mid and endpoint normalisation were the same as for the seed oil systems, due to the majority contribution from the seed oil within each impact category.

Whilst cultivation of the oilseeds provided the largest source of impacts by far, the use of power at the various stages within the process, whether it be for processing of the seed oil, production of packaging glass or use at the mayonnaise production facility, also generated a prominent environmental burden. When assessed using endpoint LCIA, these impacts manifested within the fossil depletion category which was the second most prominent category when viewing normalised data.

For the midpoint analysis, the power driven impacts arose within the natural land transformation category, which was again the second largest impact category irrespective of which oil was used for the mayonnaise formulation.

This was also true for the CFP of mayonnaise which was found to be 2.96 tCO_2eq per tonne of packaged, palletised sunflowerseed oil mayonnaise and 2.29 tCO_2eq for mayonnaise produced with rape seed oil. Whilst the seed oil contributed 69.3% and 66.4% of the impacts for the sunflower and rape seed oil mayonnaise respectively, the use of power provided between 12.8% and 17.3% of the burden from seed oil. Furthermore, power consumption at the manufacturing plant was the third largest contributor to CFP, contributing 0.3 tCO_2eq per tonne of packaged and palletised mayonnaise, representing a 9.7% contribution to the sunflowerseed oil system and a 8.8% contribution to that of rapeseed oil.

Since a perceived benefit of the novel oil-body emulsion route for production of mayonnaise was to reduce energy consumption via less intensive processing and the removal of a processing step, it was clear therefore that whilst the majority of the impacts were borne out of the cultivation of seed, innovations that reduced the overall energy consumption would also impact positively on the environmental profile of the mayonnaise.

The potential for targeted reductions in this area can be seen within figure 6.4.1, in which the middle segments of each impact bar show the impacts borne out of the process areas that could be influenced by the novel process.

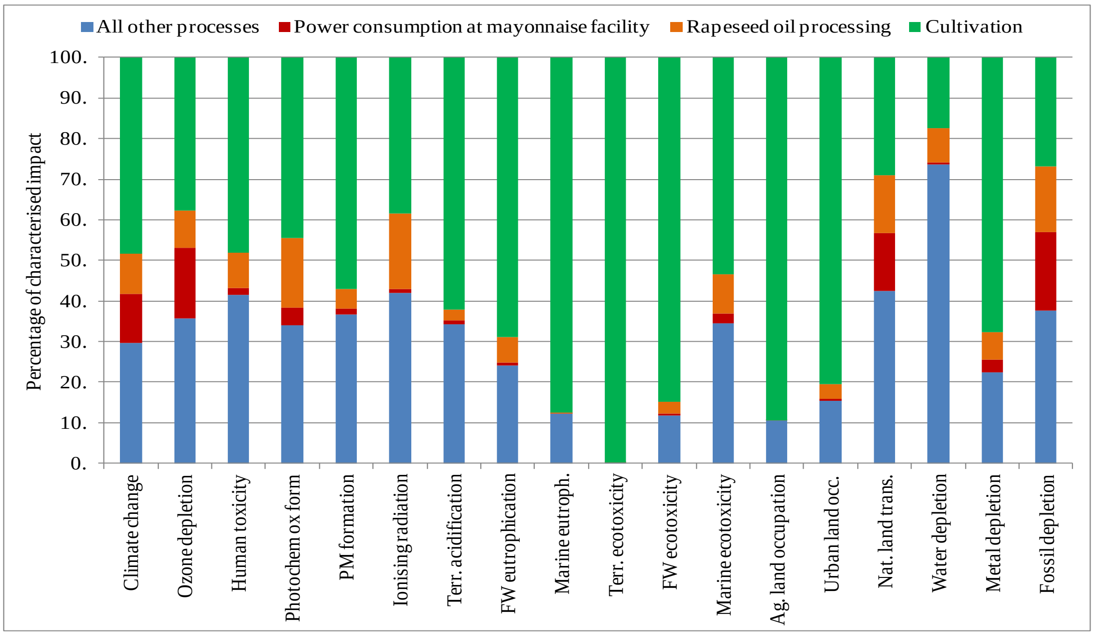

Figure 6.4-1: Rapeseed mayonnaise, percentage contributions to characterised midpoints showing contributions targeted by OB route

Similar targeted reductions could also accrue through the removal of egg from the novel oil-body emulsion. The PMF impacts for both types of mayonnaise ranked as fourth when normalised using endpoint normalisation and whilst the seed oil and packaging glass again provided the largest impact contributions, 10.5% were derived from egg usage.

Climate change did not feature as the most prominent impact category for either mayonnaise type whether analysed using midpoints or endpoints, featuring as third and fourth in endpoint analysis, and twelfth within the midpoint analysis. Whilst the prominence of climate change impacts could be artificially deflated through the choice of reference year, as discussed in chapter 5, these results could also indicate that to focus attention on GHG reductions would have the potential to cause other prominent areas of environmental impact to be missed. It could be concluded that the generation of and reliance on CFP data could potentially lead to decisions being taken that may not have the best environmental outcome.

However, cultivation of the seed provided the greatest contribution to the climate change impact category results and was also indicated as most prominent within the wider LCA results. As such, focussing attention on the greatest impact generator as determined through CFP would lead to activities that would support impact reduction in the most prominent areas within the wider LCA. Thus, whilst climate change was not the most important factor within the full LCA, the actions that would be required based on the results of the CFP alone would be beneficial to other impact categories identified as important within the overall full LCA.

Whilst seed oils are by far the largest contributor of impacts, if the recipient of CFP data began targeting areas beyond the seed oil, the CFP would indicate packaging glass and power consumption as the next most prominent areas to target for impact reduction. These findings were supported by the data from endpoint analysis within the full LCA where packaging glass and power were both prominent contributors to the fossil depletion category, which was the second most prominent category when reviewing normalised data. Thus again, whilst not directly focussing on the most prominent impacts, any actions taken as a result of data from the CFP would be beneficial to the most prominent impacts identified through full LCA.

6.5. CONCLUSIONS

In line with the overarching aim of the research, to determine whether the environmental loads of the novel production route for edible oil emulsions are lower than those of the existing processing route, mayonnaise was chosen as the case study food product for analysing the performance of the oil-body (OB) material. To enable a comparison to be made with current technology, it was necessary to generate LCA data for mayonnaise production using conventional techniques with an FU that could be replicated for analysis using OB material.

As detailed within this chapter, the cradle to gate environmental burdens of the mayonnaise system have been fully identified using both midpoint and endpoint category indicators. Furthermore, the CFPs were extracted from the full LCAs and identified as 2.96 tonnes CO_2eq when produced with sunflowerseed oil and 2.69 tonnes CO_2eq for the rape seed oil variety. This data can all be utilised for comparison with the 'mayonnaise-like oil-body emulsion' results generated within chapter 8.

It was identified through the analysis presented here that when analysing the mayonnaise cradle to gate production system, CFP and full LCA data yield consistent results with regards to the most beneficial areas for targeting to reduce environmental impacts. In this instance, the use of the single issue LCA variant would not lead to burden shifting within the system.

From the analysis conducted, it was evident that the lion's share of the contributions within all impact categories for the production of finished product comes from the use of the seed oils. As such, the largest opportunities for improving the environmental burden lie with improving the environmental profile of the oil used. Whilst the research on which this thesis is based does not intend to change the environmental profile of the cultivation of the seeds, it aims to have a positive impact on the oil production process through reduced energy and chemical usage for extraction of the oils and generation of the required emulsion.

The level to which the perceived benefits come to fruition using oil-body material will be investigated fully within chapter 8, by using the material presented here as a comparison.

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CHAPTER 7. LCA OF OIL-BODY PRODUCTION

To this point in the thesis, the case study work and analysis has concerned the evaluation of environmental impacts of existing technologies, which have been required to develop a base-line against which the process route for generating food grade emulsions from aqueous extracted oil-bodies can be compared. This chapter will outline the first stage in generating the environmental profile for the novel technology route, by detailing the case study work required to fulfil the third objective, to build an LCA model to enable quantification of the environmental impacts of aqueous extraction of oil-bodies from rapeseed and sunflower seeds.

This work will generate the following outputs:

a. Quantification of the impact on LCA results of utilising different modelling strategies for a novel technology that is still at the lab-scale
b. Identification of the environmental loads, including the carbon footprints for aqueous extraction of oil-bodies from rape and sunflower seeds
c. Scrutiny of the extent to which carbon footprint and LCA data would result in consistent decision making for progression of the novel process.

It should be noted that within the SEIBI project, the research into lab-scale aqueous extraction of oil-bodies from rape and sunflower seeds was in this instance performed by PhD researcher A. Khosla, from the University of Nottingham, Division of Food Sciences. Unless indicated through citations, the specific process description outlined within this section is based on work generated within the auspices of the SEIBI project.

7.1. RELEVANT LITERATURE

Whilst several articles had been published since the mid 1960's concerning investigations of different types of oil-bodies, interest as indicated by publication steadily grew as depicted in figure 7.1-1, after the first published works on their extraction in the early 1990's. Such works included Tzen et al. (1992) who published work on oil-bodies isolated from flax, sunflower and sesame seeds and Tzen and Huang (1992) who went on to analyse oil-bodies isolated from maize, rice, wheat, rape, soybean and jojoba.

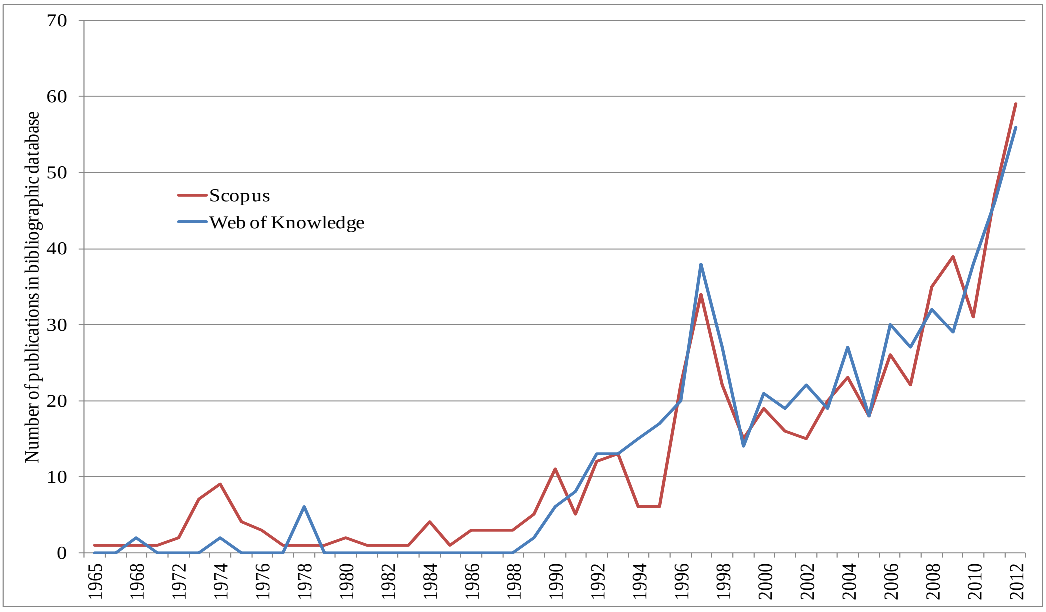

Figure 7.1-1: Analysis of number of publications concerning "oil-bodies" from main bibliographic databases

As stated within chapter 2, several papers have outlined techniques for the aqueous extraction of oil-bodies (OB) from oilseeds to generate a natural oil-in-water emulsion, of which Karkani et al. (2013) cites five (Tzen et al., 1993; Iwanaga et al., 2007; Kapchie et al., 2008; White et al., 2008; Nikiforidis and Kiosseoglou, 2009) that indicate that the processes involved may be more advantageous than tradition oil extraction techniques in terms of safety, through the elimination of the need for organic solvents and reductions in environmental pollution through the solvent-free process that would be anticipated as less energy intensive. Despite this, no analysis of the environmental credentials of such an aqueous extraction process has been published to date.

During the aqueous extraction process, which typically involves physical homogenisation or enzyme assisted digestion of the seed cell wall, the soluble cellular materials from the seed dissolve, allowing the release of oil into the bulk liquid phase. The oil can then be recovered from this phase by centrifugation resulting in a natural oil-in-water cream emulsion.

Physical OB extraction commonly consists of homogenisation using a blender (Tzen and Huang, 1992; Fisk et al., 2006; White et al., 2008; Nikifordis and Kiosseoglou, 2009), with the homogenisation medium depending on researcher and type of seed. Enzymatic extraction can be adopted as an extraction process alone or as a pre-treatment process followed by a physical extraction step (Campbell and Glatz, 2009). Whether physical or enzymatic means are used, the OBs are isolated by first filtering the homogenate, then centrifuging the filtered material to liberate the oil-bodies within a cream, which is a naturally occurring emulsion (Adams et al., 2012).

7.2. CHALLENGES WITH ASSESSMENT OF LAB-SCALE PROCESS

In order to conduct an LCA study, one must gather inventory data which as shown in the preceding chapters, is typically industrial data from established processes. As noted by Sonesson et al. (2010) a characteristic of novel systems is that no real production data exists; it is by definition, a new system and the acquisition of data is therefore problematic. Whilst primary data can be accessed by collecting mass and energy data from the lab-scale production process, such processes do not entail the same level of complexity as commercial / industrial scale processes or indeed the same requirements for processing equipment such as material and heat transfer equipment (at the minimum). Apart from the obvious difference in scale, laboratory production is also most often completed as a batch process with prominent impacts on energy consumption for start-up and shut down, in addition to potential product wastage through clean-down of equipment. Furthermore, commercial scale processes will certainly entail the use of alternative processing equipment more suited to larger scale production.

Within the lab-scale aqueous OB extraction process for example, the initial process for extraction of OB material performed by A. Khosla was to use de-hulled sunflowerseeds and homogenise them with a 0.3M (0.3 Molar) sodium bicarbonate ($NaHCO_3$) solution for 2 min using a bench blender (KRUPS Prep Expert 7000). The seed slurry was then filtered under vacuum through three layers of cheese cloth and the filtrate centrifuged at 7500 rpm using a Beckman J2-21 centrifuge (fixed rotor JA-10) for 20 minutes at 4˚C. The crude oil-body cream (COB) was collected by skimming off the top OB rich pad using a chilled metal spatula and drained on a filter paper bed. Any attempt to generate an LCA using data collected from a process with this level of manual intervention, together with the use of bench scale equipment would entail unacceptable levels of uncertainty and reduced credibility.

Additionally, lab-scale processes may exhibit a far lower yield than would be possible in a full scale facility. For example researchers within the DTI-funded 'SIPOS' project, aiming to produce oil-bodies for personal care products, used the initial extraction route described above for production of sunflower OB, during which they observed a lab-scale dry basis yield of approximately 10%. When transferred to pilot-scale for further testing however, the larger scale equipment was able to attain dry basis yields[1] in the region of 80%. Clearly, such a large discrepancy in the basic mass balance data would have an enormous impact on the overall results of an LCA, and the assessment of viability of the process.

[1] Dry basis yield refers to the yield of oil within the oil-body material compared with that theoretically available from the seed. This will be discussed further in chapter 8.

Sonesson et al. (2010) state that data for novel systems can be obtained either from lab-scale experiments, computer modelling, or a combination of the two. Within this research, a third method for data acquisition for the novel process was utilised, with LCA models being created initially using data from larger scale laboratory production and then using a commercial scale projection of that process based on mass balance data obtained from the laboratory trials.

The utilisation of the lab-based mass balance data provided a basis for selection of the main parts of equipment that would be required for such a process and the ability to compare the LCAs developed using the two methods for data acquisition would provide a valuable insight into the scale of issues surrounding LCA of processes at their early stage of development. Hetherington et al. (2014) (Appendix A), explore this further through an examination of similar issues when using LCA on lab-scale systems within different technologies.

7.3. SYSTEM DEFINITION

Based on the challenges described in the previous section, several LCA models were required in order to reduce the levels of uncertainty as far as possible within the modelling of the oil-body extraction system. Both the laboratory and commercial scale processes will be described within this section, with the data collection outlined in the following section. A summary of the LCA models developed will then be provided in section 7.5, prior to reporting the results.

7.3.1. Lab Scale production

Initial trials were performed (Khosla, 2010) to generate oil-body emulsions using similar techniques to those described within literature (Tzen and Huang, 1992; Fisk et al., 2006; White et al., 2008; Nikifordis and Kiosseoglou, 2009). This involved process steps as previously outlined in section 7.2, and depicted in figure 7.3-1, which also indicates the mass balance attained within that that process route. Sodium bicarbonate was chosen as the pre-soaking medium by the Nottingham research team, following a series of experiments to test the stability and yield of OB using a variety of acid and alkali media in addition to water. Used as a 0.3 molar solution, it provided the greatest OB stability and the highest yield (Gray, 2010).

Since the production quantities involved were so small and the level of manual intervention was so great, no attempt was made to generate an LCA based on process data acquired at this level however, due to complete lack of comparability of such data with the reference system of traditional commercial processing, as discussed in section 7.2.

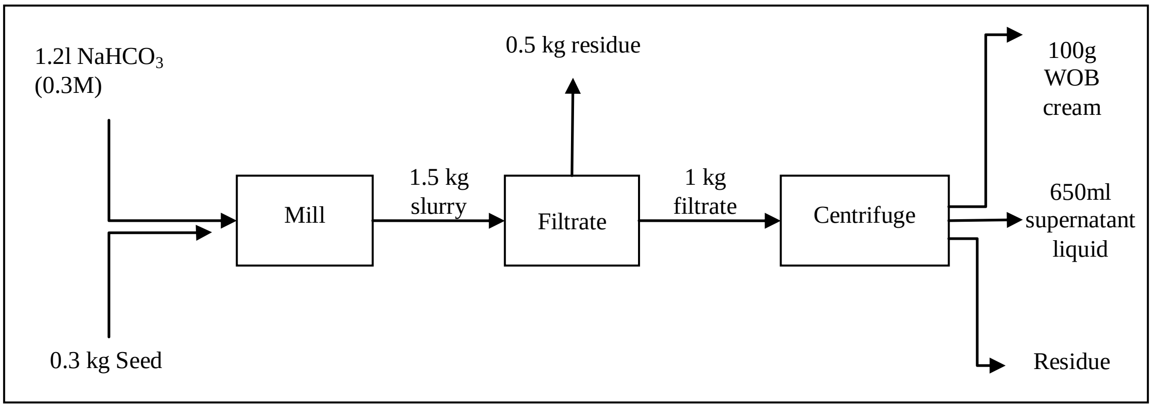

Figure 7.3-1: Schematic flow diagram for initial lab-scale production of WOB (Wet oil-bodies)

Having proved the viability of the process at the basic lab-scale, production trials were performed at the laboratories of Nottingham University School of Food Science, Sutton Bonington (SB) to increase the scale of production for sunflower oil-bodies. This enabled larger quantities of material to be generated as required for the variety of product tests necessary before use within larger scale food production trials at the premises of one of the SEIBI consortium partners.

The larger processing volumes were achieved through using a Roboqbo multifunctional cutter which was able to process 10 kg of material per batch. The Roboqbo had the advantage that it was completely self contained, with an integral jacket for heating and cooling, automated cleaning processes and the ability to process under vacuum conditions if necessary. In addition, from a data collection standpoint, it was straightforward to take power readings to record the energy consumed at each stage of the process.

The process therefore entailed the following basic steps, which are also shown in figure 7.3-2:

- Washing of the seed in peroxy peracetic acid (PPA) to protect against microbial growth
- Soaking of the seed in a 0.3 Molar solution of sodium bicarbonate (2 kg of seed with 8 litres of bicarbonate solution)
- Blending the 10 kg feed material to a slurry using the Roboqbo mixer
- Manual filtration of the slurry
- Centrifugation of the filtrate.

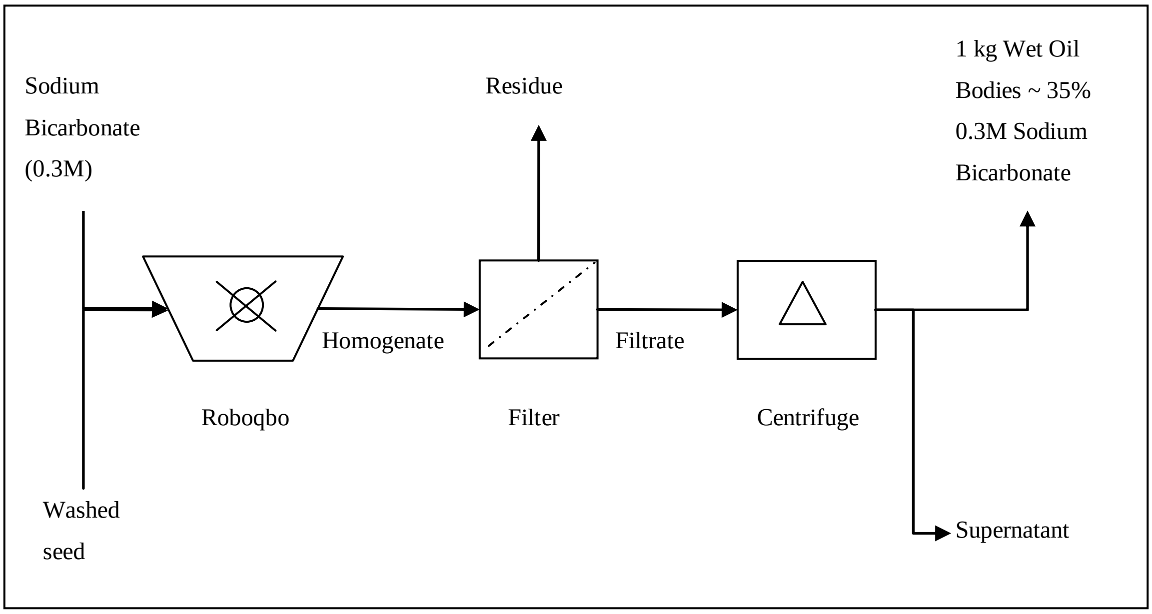

Figure 7.3-2: Process flow of production trials at Sutton Bonington

In addition, as noted in section 7.2, since the production was performed as a batch process, the Roboqbo ‘mill’ needed to be prepared for start-up, cleaned down between runs and fully cleaned after completion of the trial.

For development of the LCA of this system, a mass balance was created based on the observed quantities of materials used within the trial, as depicted in figure 7.3-2. The FU was then set as ‘production of 1 kg Wet Oil-Body cream for use as a food ingredient’ and the system boundaries were set as containing only those processes shown in 7.3-2. This FU was used to develop an LCA using entirely lab- based data, which could be compared against the LCA developed using a commercial-scale projection of that data.

7.3.2. **Commercial scale projection**

The utilisation of the mass balance data from the laboratory trials provided a basis for selection of the main parts of equipment that would be required if such a process were developed at a commercial scale. As previously stated, the ability to compare the results from LCAs developed using the two methods for data acquisition would provide a valuable insight into the implications of using different data modelling methods when developing LCAs for processes at their early stage of development.

Basing the analysis on an industrial scale unit necessitated the specification of the proposed output of the unit, such that suitable equipment could be chosen from which the manufacturers design data regarding energy usages could be used. FAO/EBRD, 1999, Agribusiness Handbooks, vol. 2, for Sunflower / Crude and Refined Oils, provided indications on the scale of sunflower oil processing facilities ranging from 50–1000 tonnes/day. Assuming that oil-body production facilities would not be required to achieve quite such high tonnages as large scale oil processors, the lower figure of 50 tonnes/day was chosen as the design basis and a mass balance was constructed on that basis.

The FU chosen was therefore ‘Production of 1 tonne Wet Oil-Body material, within a 50t/day industrial unit, for use within the food industry’, which could then be used as comparator for the LCA built using laboratory data. Whilst the mass specified for this FU differed from the lab scale FU, which was set as 1 kg, the 1 tonne was deemed more realistic for commercial conditions, however this was taken into account when comparisons with the other systems were made.

The commercial projection was identified as the most representative model to use for ongoing analysis and the FU and system boundaries were therefore modified slightly to bring them more into line with those used for the seed oils. The FU was therefore set as ‘Production of 1 tonne food grade Wet Oil-Body material, produced within a 50 t/day industrial unit, delivered to food factory’. The extended scope of this FU required that transportation from oil-body processor to food factory was included, in addition to a pasteurisation step that would replace the PPA washing process performed at the laboratory scale.

7.4. DATA GATHERING

From the process descriptions provided in the previous sections, it was clear that with four primary inputs to the process, the data required for the LCA of the OB system would be considerably less than for the oil extraction, refining and emulsification processes that it aimed to replace. Primary data for mass and energy were collected through laboratory trials or through estimation as detailed in 7.4.1.

7.4.1. Mass and Energy Data

The mass balance acquired through lab-scale processing was used to provide the quantities of the three material inputs to the system; seed, water and sodium bicarbonate. The ratio of these substances had been established following a series of laboratory trials aimed at maximising OB stability and yield. Since the properties and geometry of the seed would remain the same as the process was scaled up, it was assumed that the ratio of bicarbonate solution to seed would also remain the same, which entailed a linear scale-up of mass-balance data.

No available LCI data could be found for the PPA that was used within the process to wash the seed. It was noted however that this step was purely required for the laboratory process as protection against microbial growth would be performed differently at a commercial facility through the introduction of a pasteurisation step immediately following production as will be discussed further in this section. PPA was therefore excluded from all of the initial LCA models.

The quantity of energy required for the system was acquired either through energy monitoring within the Food Technology laboratory in Sutton Bonington or via equipment manufacturers information, using the mass balance data to determine the size of mill and centrifuge required.

Through using mass balance data collected within the laboratory trials, together with the design basis of 2.08 tonne/hour, it was determined that the mill required would need to be sized for an approximate throughput of 30,000 – 40,000 litres / hour. FrymaKoruma mills were recommended as suitable for the application by the research team at Nottingham University and a FrymaKoruma colloid mill model MZ250, with a stated capacity of 8000 – 40000 litres/hour was therefore specified for the process. The manufacturer’s specification brochure stated an energy consumption of 60 kW for this mill. For the centrifugation stage, this again needed to be sized for the duty required and therefore a Broadbent decanter centrifuge, model 750 was specified, with stated power consumption between 55 and 135 kW.

As previously stated, the power requirements of all equipment for the lab processes were measured during WOB production trials. During data collection it was noted that for the Roboqbo ‘mill’ the power required for cleaning and start-up was prominently higher than that required for the steady state operation. Consequently an additional modification to the LCA model was performed such that the results could be viewed both including and excluding the start up requirements of the Roboqbo as will be discussed later in this chapter.

LCI data for the seed, electricity and water were all taken from the Ecoinvent database available within SimaPro as previously utilised for the seed oil and mayonnaise LCA models. As before, the cultivation data for both seed types was taken as an aggregate cultivation dataset, based on the results discussed within chapter 5.

7.4.2. **Sodium bicarbonate**

Whilst sodium bicarbonate ($NaHCO_3$) is widely used within the food industry and an everyday household item, no LCI dataset could be found in any of the literature, freeware LCA databases, Ecoinvent, or any of the other available databases within SimaPro. Having failed to acquire any data specific to $NaHCO_3$ via enquiries to industry and other LCA data providers, a dataset needed to be developed.

Sodium bicarbonate is an intermediate in the production of soda powder (sodium carbonate), for which there were three datasets available within the Ecoinvent database supplied within SimaPro. However, Thieme (2000) states that because of the content of ammonium salts in the intermediate bicarbonate, a product that satisfies the quality requirements of consumers (mainly in the food industry) cannot be obtained either by drying the crude intermediate or through re-crystallisation.

Therefore, sodium bicarbonate must be produced using an aqueous solution of soda powder, which is obtained either by dissolving calcined soda ash or by decomposing crude bicarbonate with steam. This is then filtered and carbonated with pure concentrated carbon dioxide, with the heat of reaction removed by cooling. As carbonation proceeds, the sodium bicarbonate precipitates and is recovered by centrifuging followed by drying with hot air. This follows the reaction as in eq 7.4-1.

$$Na_2CO_3 + CO_2 + H_2O \rightarrow 2NaHCO_3 \qquad \textbf{Eq: 7.4-1}$$

In order to develop the bicarbonate dataset, the suitability of each of the potential sodium carbonate datasets needed to be assessed to determine which was the most appropriate for use. Sodium carbonate (soda powder) is predominantly obtained either through the Solvay process, the modified Solvay process, designed to produce ammonium chloride, or from trona ore. Trona is a hard, crystalline material from which virtually all soda powder has been produced within the US since World War Two (Thieme, 2000). The three datasets available within SimaPro represented each of these three production routes, as shown in table 7.4-1.

Table 7.4-1: Ecoinvent unit processes assessed for use as sodium bicarbonate proxy

Ecoinvent unit process name	Description	Reference
Soda, powder, at plant/RER	Soda powder (sodium carbonate) produced within a traditional Solvay Process. Based on data from two German manufacturing plants, supplemented with Finnish industrial data.	Althaus et al. (2007)
Soda, powder, at plant/US	Soda powder (sodium carbonate) produced through extraction from Trona ore (data generated 2004)	U.S. Life Cycle Inventory Database, 2012
Sodium carbonate from ammonium chloride production, at plant /GLO	Sodium Carbonate produced via modified Solvay Process in Europe. Data generated 2007, based on stoichiometric calculations.	Sutter (2007)

Althaus et al. (2007) state that 65% of global soda production is via synthetic production, inferring that the remainder is acquired through ore extraction, whilst Thieme (2000) specified the proportion produced using the modified Solvay process as 4.7%. In order to use the most representative dataset for the generation of the bicarbonate LCI, it was therefore decided to develop an aggregate dataset using the three available for soda powder using contributions of 35% from the US Soda powder (trona) set, 5% from the modified Solvay set and 60% from the Soda powder (RER) dataset.

The impact of using this data-set, rather than one of those from the databases available within SimaPro can be seen in figure 7.4-1, which shows the normalised endpoint results when each of the carbonate datasets were analysed within SimaPro.

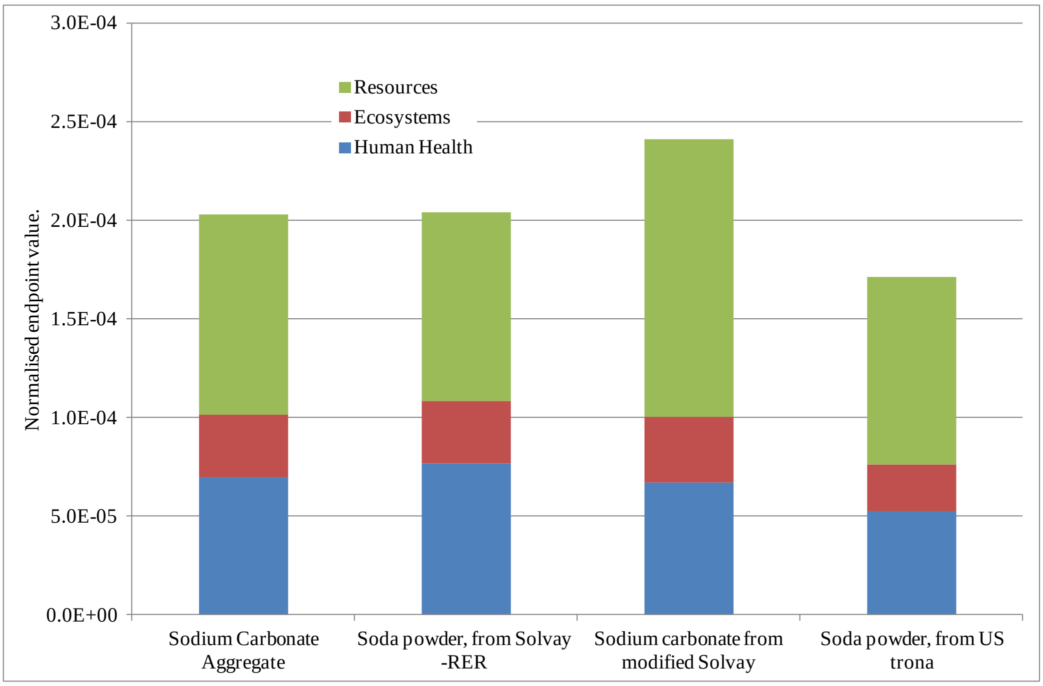

Figure 7.4-1: Results of the LCIA performed to compare aggregate Soda Powder LCI with its contributing parts

On reviewing the datasets in detail, it was determined that the two synthetic processes utilised different allocation methods to partition the data between by-products produced. The Soda powder (RER) dataset used economic allocation (33% to soda powder and 67% to calcium chloride) whilst the modified Solvay set used stoichiometric allocation, between the soda powder and its co-product ammonium chloride.

It was decided to use stoichiometric allocation as the method for both processes as ISO 14044:2006 states that physical allocation parameters are generally more preferable than economic and unlike the case for the seed oils, no positive justification of the use of economic based on product value could be found. The inputs within the Soda powder (RER) dataset were therefore modified accordingly, based on equation 7.4-2, where the molar masses required are provided below each substance. Thus the proportion of inputs and outputs attributed to the soda powder was increased to 48.85% (106 ÷ (106 + 111)).

$$2\,NaCl + CaCO_3 \rightarrow Na_2CO_3 + CaCl_2 \qquad \textbf{Eq: 7.4-2}$$
(117 g) (100 g) (106 g) (111 g)

The aggregate dataset for sodium carbonate was therefore produced, and this was utilised to generate an LCI for the production of sodium bicarbonate by using the stoichiometry of equation 7.4-3, which again gives the required molar mass below each substance.

$$Na_2CO_3 + CO_2 + H_2O \rightarrow 2NaHCO_3 \qquad \textbf{Eq: 7.4-3}$$
(106 g) (40 g) (18 g) (168 g)

This resulted in a dataset that incorporated the material inputs shown in table 7.3-4 to produce 1 kg of sodium bicarbonate:

Table 7.4-2: Material input data for generation of sodium bicarbonate LCI

Substance	Amount (kg)	Dataset name	Source
Aggregate Soda powder	0.6310	Sodium Carbonate for bicarb production	Self generated for this study
Carbon dioxide	0.2381	Carbon dioxide liquid, at plant/RER U	Ecoinvent dataset: Althaus et al. (2007)
Water	0.1071	Process water, ion exchange, production mix, at plant, from groundwater RER S	ELCD dataset: http:/lca.jrc.ec.europa.eu (2010)

The sodium bicarbonate used within the oil-body process is in the form of a 0.3 Molar (0.3 M) solution used for pre-soaking of the seed which is discarded after OB collection. Whilst any future optimisation of the process may introduce bicarbonate recovery as part of the process, for the purposes of the modelling, the treatment in the commercial scale LCA was assumed to be the same as that of the lab scale process i.e. no bicarbonate recovery.

It was therefore necessary to determine the mass of $NaHCO_3$ powder required for the solution. A 0.3 M solution requires 0.3 moles of the substance in question per litre of water. As the molar mass of $NaHCO_3$ is 84 g/mol, a 1 litre solution would therefore require 25.2 g of $NaHCO_3$ powder per litre of water (0.3 x 84). This quantity was then factored up as required within the mass balance data.

7.4.3. **Power requirements for pasteurisation**

Since the PPA washing stage of the laboratory process would be replaced with a pasteurisation unit within a commercial facility, the LCAs that would be used for ongoing analysis and comparison with existing technologies required the power requirements of such a pasteurisation unit to be included.

The industry most widely associated with pasteurisation technologies is the milk industry, elements of which, as previously noted in section 7.1 have been subject to several LCA studies in the past. A search was therefore performed for relevant pasteurisation data within a dairy setting, for which a suitable dataset was found within the Danish LCAFood database. This data indicated energy inputs of 50 kWh/t of milk for heat energy and 54 kWh/t of milk for the electricity requirements.

Data available via chapter 3 of FAO (1992) concerning the energy requirements for milk processing, also provided data for the pasteurisation stage. Here it was stated that pasteurisation of each tonne of milk within a simple milk processing plant would require 180 MJ (50 kWh) of heat, a figure that agreed with that supplied by the LCA Food data. It also stated that 90 MJ (25 kWh) of electricity was required for ancillary processes such as *'pumps, stirrers, refrigeration plant compressors and various servomechanisms.'* . a figure that was somewhat lower than the 50 kWh supplied by the LCA Food data.

The figure of 50 kWh was therefore used for the heat requirements of the pasteurisation process and 54 kWh used for the electricity requirements as the worst case scenario, with an analysis also performed at 25 kWh electricity to check for sensitivity.

7.4.4. **Transport**

For the generation of the rape and sunflowerseed oil-body process models at the commercial scale, it was necessary to include the transportation that would be required within the system, in the same way that it had been included for the seed oil LCAs.

Since the same cultivation datasets were being utilised as had been used for the seed oils, it was appropriate to use the same transport distances. Therefore although aggregate cultivation data was used, it was assumed that the sunflowerseeds were transported from a farm in Spain to an extraction facility in Spain for extraction of the oil-bodies, which were then transported to the Netherlands. This entailed both road and sea transport as will be detailed in table 7.5-1. For the rapeseed oil-body system, it was assumed that the seed was transported by road within Germany from the farm to the oil-body extraction facility, prior to road haulage to the Europort in Rotterdam. In both cases, the wet oil-bodies were then transported via sea and road to the UK for food use. Full details of the distances used within the models are provided in table 7.5-1 of the next section.

7.5. LCA VARIANTS

Based on the mass and energy balances described above, five models were therefore developed on the following basis:

i. Sunflower OB laboratory production – trial data including start-up conditions
ii. Sunflower OB laboratory production – trial data excluding start-up conditions
iii. Commercial scale projection of the sunflowerseed OB production – using only lab process stages
iv. Commercial production of sunflowerseed OB including pasteurisation and transport
v. Commercial production of rapeseed OB including pasteurisation and transport

Table 7.5-1: Details of input data for the five LCA models used to analyse OB system

LCA input process	Units	Lab scale Whole run: Sunflower (1 kg FU) (LCA - i)	Lab scale run only : Sunflower (1 kg FU) (LCA - ii)	Commercial scale – lab processes: Sunflower (1 tonne FU) (LCA - iii)	Commercial Rapeseed OB: Modified FU (1 tonne FU) (LCA - iv)	Commercial Sunflowerseed OB: Modified FU (1 tonne FU) (LCA - v)
Seed	kg	4.615	4.615	4615.4	4615.4	4615.4
Water	kg	18.4625	18.462	18461.5	18461.5	18461.5
Bicarbonate	kg	0.465	0.465	465.2	465.2	465.2
Power for mill	kWh	2.378	0.440	28.8	28.8	28.8
Power for centrifuge	kWh	4.792	4.792	45.6	45.6	45.6
Power for pasteurisation	kWh				54	54
Heat for pasteurisation	kWh				50	50
Transport farm to OB extractor (road)	km				65	338
Transport farm to OB extractor (sea)	km					1756
Transport from extractor to Europort (road)	km				650	
Transport from extractor to food factory (sea)	km				390	390
Transport from extractor to food factory (road)	km				196	196

7.6. RESULTS AND DISCUSSION

7.6.1. Comparative analysis of lab-scale and projected commercial scale LCAs

Having developed LCA models for both of the systems for producing sunflowerseed WOB as described in section 7.3.1 and 7.3.2, comparative analysis of the results obtained from both models was performed. The data obtained from the laboratory trials also incorporated the power required for start-up conditions and as such calculations were performed on models that included and excluded this data. Whilst the mass quantities of the FU's concerned were different (1 kg at the laboratory scale vs 1 tonne for the commercial scale), the analysis was made comparable by scaling up the impacts from the lab model by one thousand, such that both models yielded the results for production of one tonne.

The results of the LCIA performed on the three initial LCA models indicated not only that the power required for start up conditions has a considerable impact on the results of the system, but also that the contribution from the seed cultivation is dominant, with contributions of 78% for the lab-scale LCA including start-up conditions to 97% for the LCA of the commercial scale projection. As such, it was decided to re-run the analysis with the cultivation data excluded such that the impact of start-up conditions could be more clearly seen. The results of this analysis on normalised endpoints is presented in figure 7.6-1, whilst the analysis including cultivation can be seen in Appendix E.

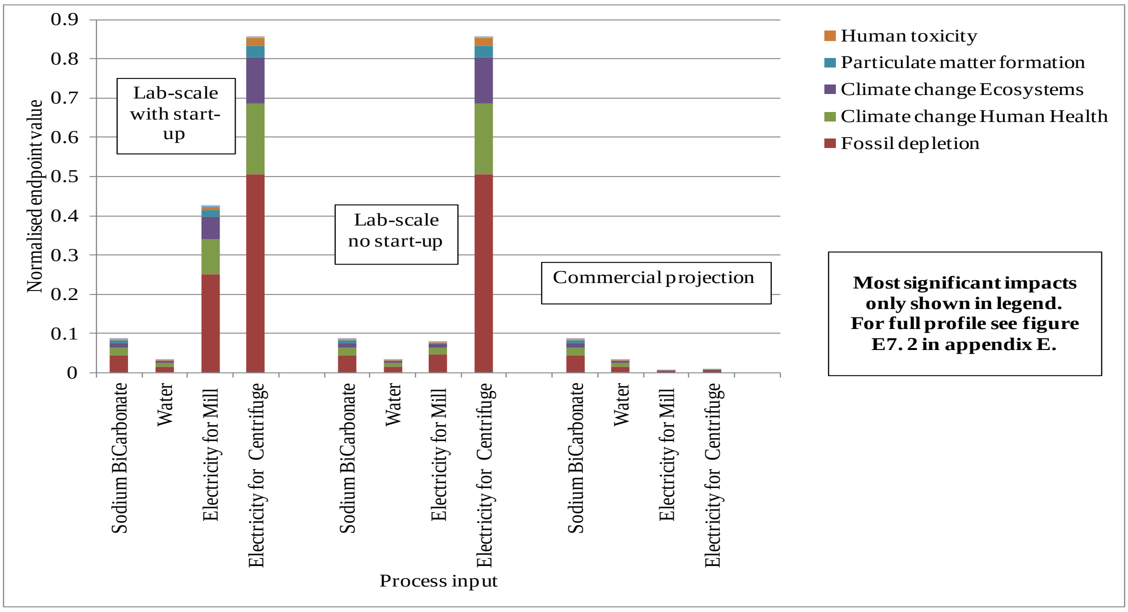

Figure 7.6-1: Comparison of normalised endpoint results for model developed with different data acquisition methods

From figure 7.6-1, it was evident that when the start-up power requirements for the milling were removed, the impacts caused by the energy consumption fell prominently. However irrespective of the steady-state or start-up requirements for the Roboqbo, it was clear that at a lab scale, the most prominent contributor to environmental load was the energy requirement for centrifugation, part of which was as a result of power required to bring the centrifuge to operating speed for each batch.

This was very different from the results obtained for the commercial projection of the novel system, where much lower values were calculated. In this case, the use of sodium bi-carbonate for pre-soaking the seed was shown as the largest contributor of environmental impacts. Apart from the efficiencies of scale afforded by the larger equipment specified for the commercial process, the power required for starting up the lab-scale centrifugation for each batch clearly had an adverse effect on the results generated.

Whether modelled using lab-scale data or commercial scale projection, the most prominent impact categories when analysed and normalised at the endpoint level were found to be fossil depletion, natural land transformation and ionising radiation. All three of these were as a result of power generation, for the electricity required for the milling and centrifugation stages, together with that required for manufacture of the sodium bicarbonate. For the fossil depletion and natural land transformation impacts, these results were borne out of the extraction of fossil fuels to be burnt for power generation as part of the electricity mix. The ionising radiation impact was due to use of nuclear power within that mix.

The data was also analysed at the midpoint level to identify whether consistent results were obtained. Once more, the cultivation of the sunflowerseed provided the dominant process contribution, ranging from 76.7% for the lab-scale data including start-up conditions to 97.3% for the commercial projection of the process. The system was therefore examined without the cultivation stage to more clearly see the contributions of the different stages, together with the most prominent impact categories for those stages. The results of this analysis can be seen in figure 7.6-2, with the results including the cultivation stage provided in figure E7.3, appendix E.

The midpoint analysis in figure 7.6-2 confirms the previous findings, that the data generated using the laboratory-scale data varies prominently from that obtained using the commercial projection of the system. Analysis of the lab-scale systems both with and without start-up conditions showed that the largest contributor of environmental impacts was the electricity for centrifugation. For the commercial scale system however, the largest contributor was the bicarbonate of soda, consistent with the findings of the endpoint analysis.

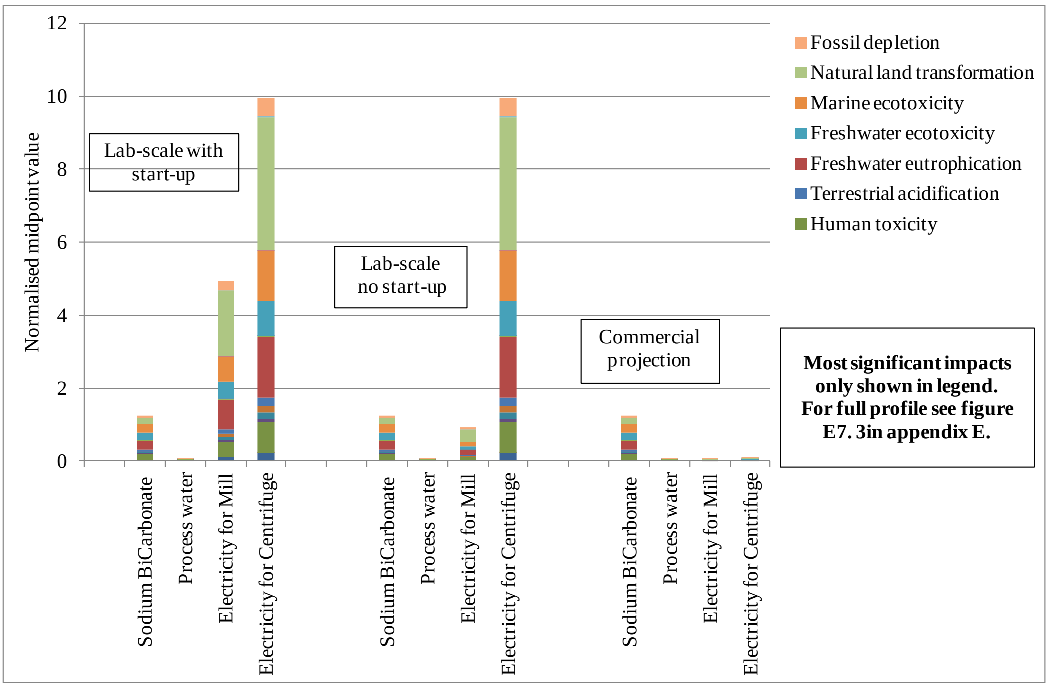

Figure 7.6-2: Comparison of normalised endpoint results for model developed with different data acquisition methods

As with the endpoint analysis, the use of fossil fuels was found to be the driver for the most prominent impacts, with exploration for such fuels causing the largest impact category natural land transformation and effluent disposal from the use of coal giving rise to the next two most prominent categories, freshwater eutrophication and marine eco-toxicity.

7.6.2. **Environmental loads of rape and sunflowerseed oil-body production**

Having examined the difference in impacts arising from using the different scale models for the sunflowerseed OB system, the commercial projection was chosen as the most realistic approximation for the system under scrutiny and LCIA was performed on models generated for both rape and sunflowerseed OB systems as outlined in section 7.5.

Normalised results of the LCIA for both WOB systems are shown in figure 7.6-3, from which it is evident that both systems have the highest levels of impacts within the Damage to Eco-systems area of protection (AoP), with the results for the Damage to Human Health and Damage to Resources AoPs being almost equal.

This environmental profile was consistent with that obtained for the conventional production of both oils, within which the impacts that arose from agricultural land transformation dominated the Eco-systems AoP. The consistency of the profile is unsurprising given that both the conventional oil system and the WOB extraction systems were dominated by the seed cultivation stages.

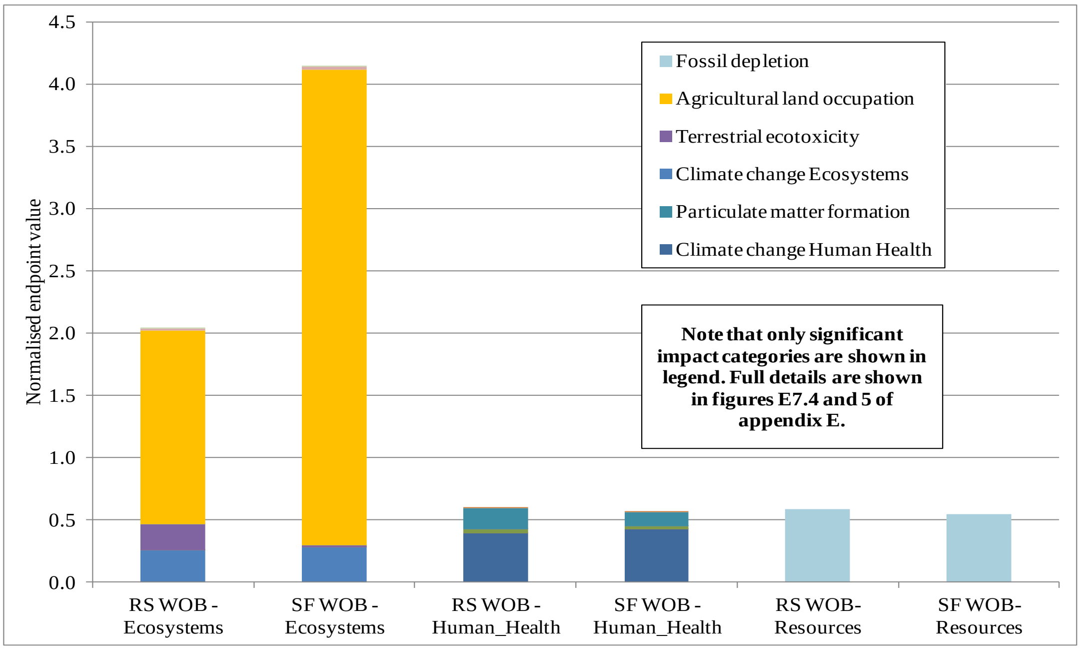

Figure 7.6-3: Normalised endpoint results for rape and sunflowerseed WOB (ReCiPe(2008)): RS = Rapeseed; SF = Sunflower

To more clearly see the prominent contributions within each of the normalised endpoints, the individual impact categories were reviewed, of which the five with the greatest magnitude for both WOB systems are shown in figure 7.6-4. Consistent with the overall assessment of endpoints, the cultivation stage dominated the contribution to each of the most prominent individual impact categories, within which the top four normalised impacts for both systems were the same with terrestrial eco-toxicity being the fifth largest impact category for the rapeseed system, whereas particulate matter formation was fifth for the sunflower system. This difference was borne out of the use of pyrethroid pesticides for rapeseed crop protection, which raised the profile of the terrestrial eco-toxicity impact within that system. For the largest normalised endpoint, that of agricultural land occupation, the magnitude of the impact for the sunflowerseed WOB system (3.82) is several times higher than that for the rapeseed WOB system, at 1.55, a feature consistent with the increased land required for sunflower cultivation as discussed in chapter 5.

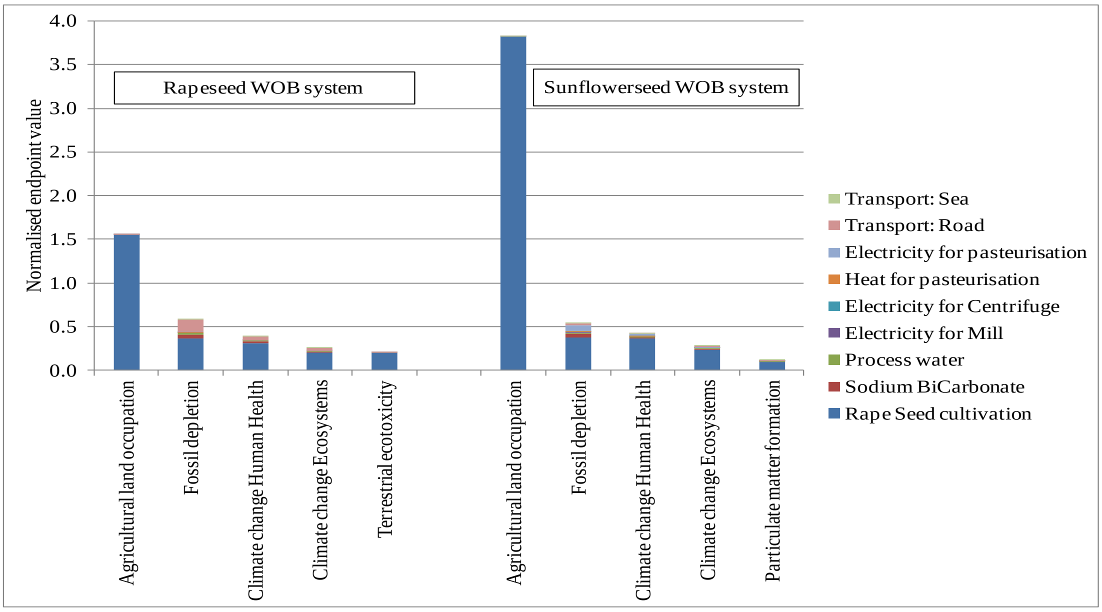

Figure 7.6-4: Endpoint LCIA for WOB production systems: Normalised endpoint values for top five midpoints ranked by magnitude

Fossil depletion, climate change human health and climate change ecosystems were the next largest impacts within both systems and had similar magnitudes for both seed types. The full profile of endpoints for the WOB systems are shown in figures E7.4 and E7.5 in Appendix E.

Whilst the largest impact categories for both WOB systems were agricultural land occupation, fossil depletion and both climate change categories when normalised at endpoint, the toxicity and eutrophication impacts were indicated as having higher relative levels when reviewing the normalised midpoint data. A summary of this analysis can be seen in figure 7.6-5 for which the top five midpoint categories ranked by magnitude are shown. The full midpoint profile can again be seen in figures E7.6 and 7.7 in Appendix E.

As with the endpoint data, since the cultivation of the seed had such an overbearing contribution within the system, the impact profile of the WOB production systems mimicked the results profile for the seed cultivation, with terrestrial eco-toxicity indicated as the largest normalised midpoint category within the rapeseed WOB system and marine eutrophication the largest normalised midpoint within the sunflowerseed WOB system.

As with the raw seed systems, climate change midpoints did not feature within the top five normalised midpoints, whereas they were prominent within the normalised endpoint results. This is as a result of the increased prominence of the toxicity impacts through midpoint normalisation as discussed in chapters 4 and 5.

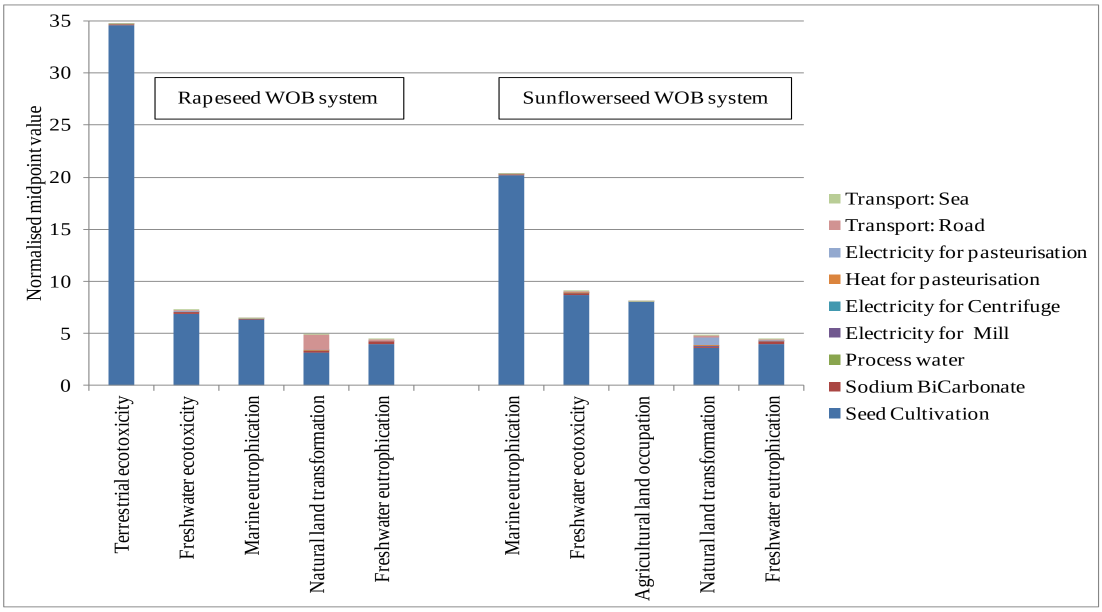

Figure 7.6-5: Midpoint LCIA for WOB production systems: Normalised midpoint values for top five midpoints ranked by magnitude.

The characterised midpoint data for both WOB production systems at the projected commercial scale are shown in table 7.6-1.

Table 7.6-1: Characterised midpoint data of commercial scale projection for OB process

Impact category	Unit	Rapeseed WOB	Sunflowerseed WOB
Climate change (CC)	kg CO_2 eq	5598	6076
Ozone depletion (OD)	kg CFC-11 eq	0.0005	0.0004
Human toxicity (HT)	kg 1,4-DB eq	951	597
Photochemical oxidant formation (POF)	kg NMVOC	18.6	23.1
Particulate matter formation (PMF	kg PM_{10} eq	13.3	8.9
Ionising radiation (IR)	kg U_{235} eq	476	553
Terrestrial acidification (TA)	kg SO_2 eq	66.7	26.1
Freshwater eutrophication (FE)	kg P eq	1.8	1.8
Marine eutrophication (ME)	kg N eq	65	205
Terrestrial ecotoxicity (TET)	kg 1,4-DB eq	284.2	26.1
Freshwater ecotoxicity (FET)	kg 1,4-DB eq	78.8	98.1
Marine ecotoxicity (MET)	kg 1,4-DB eq	20.3	18.3
Agricultural land occupation (ALO)	m^2a	14782	36289
Urban land occupation (ULO)	m^2a	143	232
Natural land transformation (NLT	m^2	0.8	0.8
Water depletion (WD)	m^3	42.1	38.3
Metal depletion (MD)	kg Fe eq	265	250
Fossil depletion (FD)	kg oil eq	960	892

Whilst reviewing the normalised results is useful for determining those impacts with higher values relative to the European norm, it is through looking at the contributions to each of the relevant characterised impact categories that levels of impact derived from each process stage can be properly established. The relative contributions from each process input to the five most prominent impact categories as identified through midpoint normalisation are shown in figure 7.6-6.

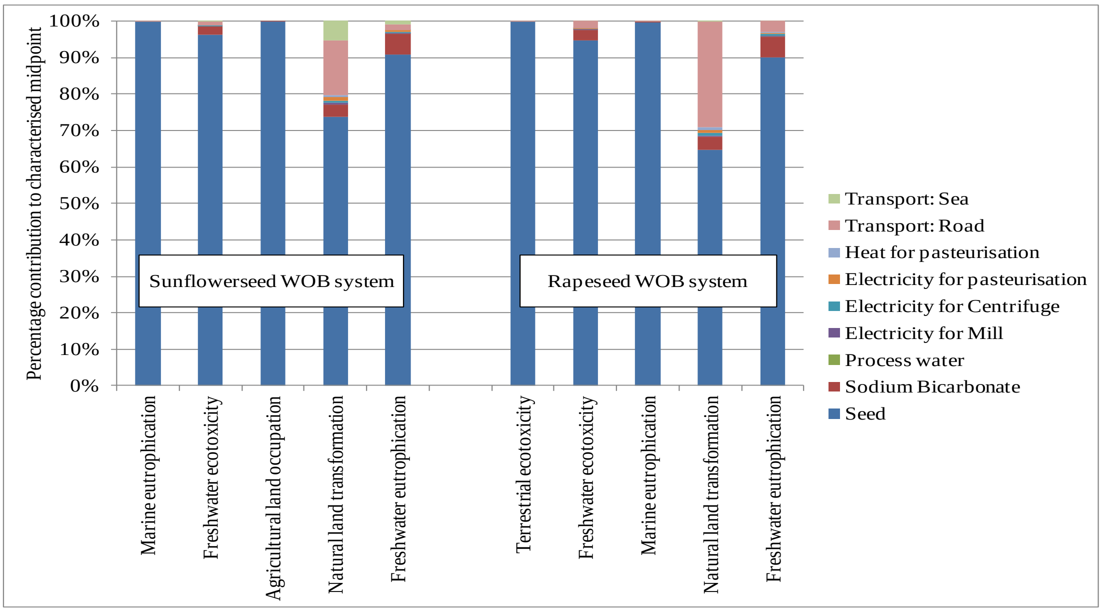

Figure 7.6-6: Contribution of each process stage to characterised midpoints identified as largest through midpoint normalisation

For each of the largest midpoint indicators, the cultivation of the seed had the greatest contribution. This ranged from a 100% contribution to impact for the top impact categories marine eutrophication and terrestrial eco-toxicity for sunflower and rape systems respectively, to contributions of 73.6% and 64.7% for the sunflower and rape WOB systems within the natural land transformation category.

The impacts derived from road transport for the seed also provided substantial contributions within this category, at 15.0% for the sunflower and 29.1% for the rape system. This was due to the land transformation necessary for the extraction of crude oil for the production of diesel required to fuel the road transport. The third largest contributor of impacts was the sodium bicarbonate used for soaking the seed, which yielded contributions of up to 5.6% for freshwater eutrophication midpoint within the sunflowerseed system and 5.7% for the same indicator for the rapeseed system.

Within both systems, the impacts derived from the electricity required for pasteurisation contributed a minimal amount to the overall system burdens, with maximum contributions of 0.7% and 0.8% within the natural land transformation category. When the power consumption for this stage was reduced from 54 kWh to 25 kWh, this reduced to a maximum of 0.4%. Full details of all process contributions can be found in tables E7.3 and E7.4 in appendix E.

As discussed in previous chapters, Van Hoof et al. (2013) advocate a combined approach whereby characterised midpoint data is reviewed for those impacts assessed as highest when normalised using endpoint normalisation. Following this approach, the five most prominent endpoints were agricultural land occupation, fossil depletion, climate change - human health, and climate change - ecosystems for both WOB systems, with particulate matter formation being the fifth for the sunflowerseed system and terrestrial eco-toxicity being the fifth for the rapeseed system. Since the two climate change endpoint impacts map to a single climate change midpoint, this yields four midpoint categories, as shown in figure 7.6-7.

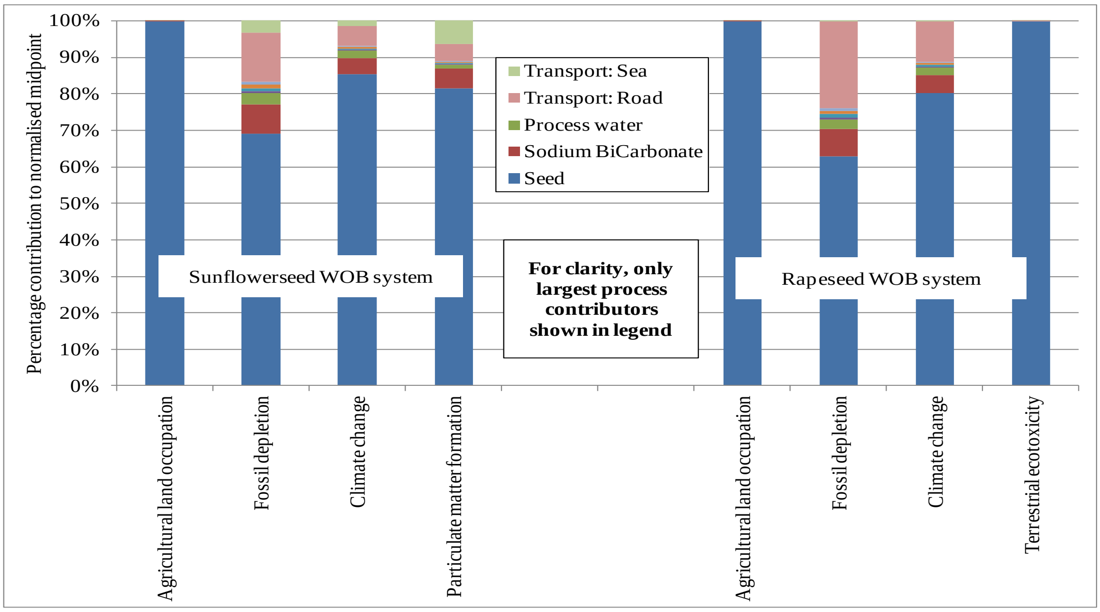

Figure 7.6-7: Contribution of each process stage to characterised midpoints deemed most relevant through endpoint normalisation

From figure 7.6-7 it is evident that whilst the seed cultivation provides the largest contribution to each impact category, transportation is once more the second largest contributor of impacts within the highest ranked impact categories, with road transport contributing 23.9% of the fossil depletion impact and 11.2% of the climate change midpoint for the rapeseed system. Within the sunflower system, the contributions from transport were slightly more modest, but still the second largest contributor with road haulage providing and 13.3% of the fossil depletion impact 5.3% of the particulate matter formation impact as shown in table 7.6-2.

Table 7.6-2: Percentage contributions to most prominent midpoint impacts as identified by endpoint normalisation

	Sunflowerseed WOB system				Rapeseed WOB system			
	ALO	FD	CC	PMF	ALO	FD	CC	TET
Seed	100.0%	69.2%	85.3%	81.6%	99.9%	62.9%	80.3%	99.9%
Sodium BiCarbonate	0.0%	8.0%	4.5%	5.3%	0.1%	7.4%	4.8%	0.0%
Process water	0.0%	2.9%	2.0%	1.1%	0.0%	2.7%	2.1%	0.0%
Electricity for Mill	0.0%	0.6%	0.3%	0.2%	0.0%	0.5%	0.3%	0.0%
Electricity for Centrifuge	0.0%	0.9%	0.4%	0.3%	0.0%	0.8%	0.4%	0.0%
Electricity for pasteurisation	0.0%	1.0%	0.5%	0.3%	0.0%	1.0%	0.5%	0.0%
Heat for pasteurisation	0.0%	0.7%	0.3%	0.1%	0.0%	0.7%	0.3%	0.0%
Transport: Road	0.0%	13.3%	5.3%	4.9%	0.0%	23.9%	11.2%	0.0%
Transport: Sea	0.0%	3.3%	1.5%	6.3%	0.0%	0.1%	0.0%	0.0%

As with the analysis of impacts relevant at midpoint normalisation, the contributions from pasteurisation power are minimal with the maximum levels being 1% for fossil depletion. When assessing the sensitivity of the system by reducing the power requirements of this stage to 25 kWh, this contribution reduced further down to 0.5% for both seed systems, indicating that further attempts at accessing more accurate data for this stage would not be productive or necessary.

7.6.3. **Carbon footprints of both oil-body production systems**

As indicated in chapter 6, since the GWP characterisation method within ReCiPe (2008) uses the IPCC (2007) equivalence factors (Goedkoop et al., 2013) and as the scope of the assessment conforms to PAS2050:2011, the climate change impact category results at the midpoint level provide the CFP for the two systems.

From the characterised data previously presented in table 7.6-1 it was evident that the CFPs of one tonne of food grade Wet Oil-Body material, produced within a 50t/day industrial unit, delivered to food factory' were 6.1 tCO_2eq for the sunflower WOB system and 5.6 tCO_2eq for the rapeseed system. As previously indicated, a comparison between these results and those generated for refined oil produced via conventional techniques was not entirely appropriate since the systems were not functionally equivalent. However whilst not directly comparable, the magnitude of difference between the results of both systems and the respective results from chapter 5, indicated that the functionally equivalent comparison to be performed in chapter 8 will almost certainly indicate that the burdens for WOB processes would be prominently higher than those for conventional processing.

The WOB with the presently used conditions and assumptions yielded CFPs that were over twice the levels calculated for the conventionally processed oils (2.6 tCO_2eq for refined sunflowerseed oil and 2.3 tCO_2eq for refined rapeseed oil), values that were born out of the greatly increased impacts derived from cultivation and transportation of the seeds. This indicates that OB yield from the seed will be a pivotal variable for the environmental performance of the system.

Analysing the CFPs of both systems further, figure 7.6-8 shows the contributions from each processing element, from which it was evident that the largest single contributor to CFP was the cultivation of the seed for both oil-body systems, contributing 85% of the impacts for the sunflowerseed oil-body process and 80% of the impacts for the rapeseed oil-body production.

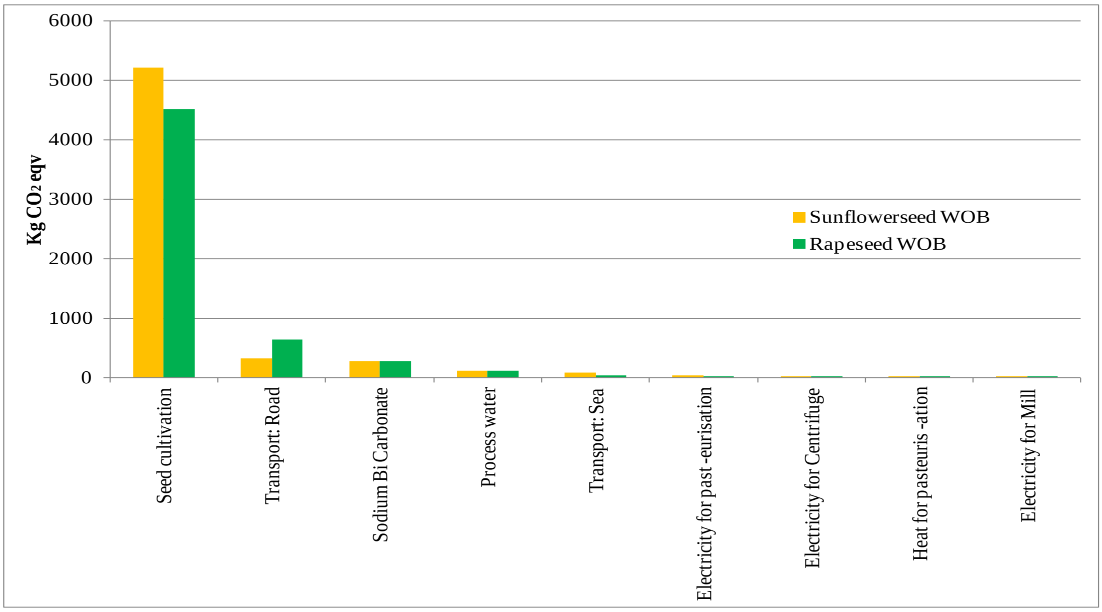

Figure 7.6-8: Relative contributions of process stages to CFP for WOB systems

The second largest contributor in both systems was the transportation by road, which contributed 5.3% of the CFP for the sunflowerseed system and 11% of the GHG emissions for the rapeseed system, with the impacts derived from the use of sodium bicarbonate causing its use to be indicated as the third highest contributor of GHGs. This information could lead improvements to be targeted at reducing the use of this material for soaking the seed, an activity which would also yield benefits across the wider spectrum of environmental impacts, with bicarbonate being indicated as a key contributor to several of the most prominent impacts, both when normalised at midpoint and endpoint level.

These results were largely consistent with those attained by the full LCA as previously shown in table 7.6-2, where the contribution from seed was the largest contributor to each of the most prominent impact categories, followed by road transport for the fossil depletion and climate change categories within both seed type WOB LCAs.

7.7. SUMMARY

The development of LCA models to analyse the aqueous extraction of oil-bodies entailed many of the same issues encountered when building such models for assessment of the conventional technologies as detailed within chapters 5 and 6. However, whilst LCA is undoubtedly an important tool for identifying the environmental burdens of any process, analysis of novel systems and processes at the early development stage such as with the OB production process, intensifies many of the challenges with LCA model creation.

Research objective 3 entailed the quantification of the CFP and wider environmental impacts of aqueous extraction of oil-bodies from rapeseed and sunflowerseeds, together with an examination of the assumptions and simplifications required to generate an LCA for the projected commercial scale application of a novel technology that is still at the lab-scale. To achieve this, a pragmatic method was developed for acquiring the necessary LCI data by converting laboratory acquired mass balance data to a commercial scale based on anticipated tonnages, and utilising manufacturers data for energy consumption.

This was developed after reviewing the results from three comparative LCAs performed on the same process, using different sources of data and modelling assumptions. These covered the production of food grade oil-bodies generated using i) laboratory measurements including energy for start-up ii) laboratory measurements with start-up energy removed and iii) laboratory mass balances projected as a continuous 50 tonne/day production unit, using manufacturers data for equipment energy consumption.

Whilst the impact categories identified as most substantial through mid and endpoint normalisation were consistent for all three models, the magnitude of contributions were considerably different. Despite removing the energy requirements for start-up of the batch production, a large disparity existed between the projected commercial scale LCA and that generated using laboratory results alone. Even taking into account the additional energy requirements for material heating and transfer processes at the commercial scale, it was clear that basing an LCA on laboratory data alone would give rise to very different conclusions concerning the environmental credentials of the process and potentially lead to ill informed decisions being made. For consistency and in an attempt to reduce the levels of uncertainly therefore the commercial projection was chosen as the LCA model to develop further.

The method used for modelling the commercial scale proxy could be applied equally well to other processes at the early developmental stage, with the step-wise approach shown in figure 7.7-1.

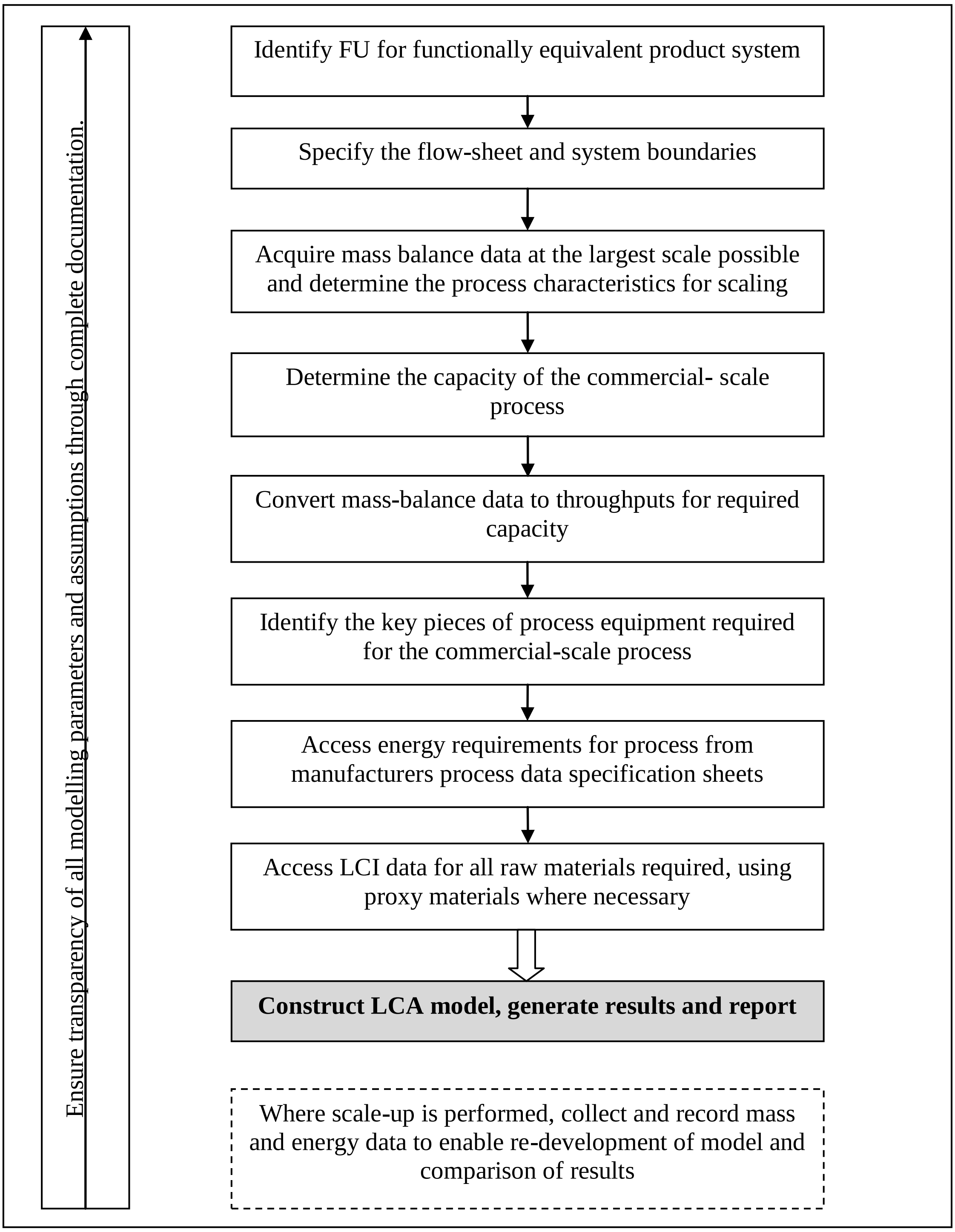

Figure 7.7-1: Schematic flow of steps required for generation of novel process LCA at commercial scale

Appropriate LCA models for the oil-body extraction process were therefore created, analysed and interpreted. In addition, results were produced for completion of objective 5, which entailed examining the use of CFP as an effective environmental indicator for this system. As with the mayonnaise LCA results, whilst reviewing the normalised midpoint values did not place climate change amongst the highest impact scores, the main contributors to the CFP were consistent with those that drove the wider impacts with the largest normalised values. As such it was clear that for this system, targeted optimisation based on CFP data would lead to improvements throughout the wider spectrum of environmental impacts and CFP could therefore be judged as an effective indicator of environmental performance.

The combined reporting approach advocated by Van Hoof et al. (2013), was again adopted whereby characterised midpoint data was reviewed for those impacts assessed as highest when normalised using endpoint normalisation. From this analysis, the midpoints reviewed were agricultural land occupation, fossil depletion and climate change for both WOB systems, with particulate matter formation for the sunflowerseed system and terrestrial eco-toxicity for the rapeseed system.

The perceived environmental benefits of the WOB process derive from reduced energy consumption during processing and the simplification of processing stages. From the analysis presented here, it was evident that with the extraction yields reported to date from the laboratory trials for OB production, the impacts borne out of these areas were dwarfed by those arsing from the seed cultivation. In addition, the poor OB yields from seed had caused transportation to take a more prominent role through the need to convey higher levels of seed to the processor.

Potential reductions in cultivation impacts could be afforded through reductions in the power requirements for seed drying, for which research ongoing at the University of Nottingham indicated that seed could perhaps be processed in a wetter state than currently possible for standard oil extraction Gray (2012). However the largest environmental gains would arise through increased OB extraction yields from the seed, which would reduce the contributions both from the seed and the transport required. The investigation of both of these process modifications will be detailed within the next chapter.

7.8. CONCLUSIONS

The objectives of the research detailed within this chapter entailed the construction of a series of LCA models to enable the identification of the CFP and full environmental profiles of commercial scale, food-grade, oil-body production from rape and sunflowerseeds. Within this, a quantitative analysis of modelling issues arising when using LCA to investigate the environmental credentials of a novel process was performed.

Having developed models for the commercial-scale production of rape and sunflowerseed oil-body materials, the CFPs per tonne of oil-body product were found to be 6.1 t CO_2eq for the sunflower system and 5.6 tCO_2eq for the rapeseed oil-body system. The wider environmental loads of both product systems were also established, from which agricultural land occupation was determined as the most prominent normalised endpoint impact for both systems, with the highest normalised midpoints being terrestrial eco-toxicity for the rapeseed oil-body system and marine eutrophication for the sunflowerseed system. Climate change impacts ranked third and fourth within the top impact categories for endpoint normalisation, however were not inside the top five normalised midpoints.

Whether assessed using CFP or a full LCA, the impacts from cultivation of seed for the oil input had a dominant contribution within the system, with power usage impacts being the next highest contributor. It was therefore evident that for this system, CFP and the fuller LCA data would result in consistent decision making for progression of the novel process.

Having successfully developed the LCA models to establish the environmental loads for the production of food-grade oil-body material from both seed types, the models were then utilised as a raw material input for the generation of a mayonnaise-like food-grade emulsion. The modelling and analysis work for this will be detailed within the next chapter.

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CHAPTER 8. LCA OF MAYONAISE-LIKE OIL-BODY EMULSION

To fulfil the primary aim of the thesis, the environmental performance of food grade edible oil emulsions produced via aqueous extraction of oil-bodies was evaluated within a functionally equivalent product system. This was achieved by using conventional mayonnaise production and an equivalent emulsion product, manufactured using rape and sunflowerseed oil-bodies.

This chapter describes the generation of a series of LCA models for the mayonnaise-like oil-body emulsions, together with the comparative analysis performed with the case-study product mayonnaise. The modelling and analysis performed fulfilled the fourth, fifth and sixth objectives of this research, by identifying the environmental loads of the production of a 'mayonnaise-like' emulsion using rape and sunflowerseed oil-bodies, evaluating the suitability of single-issue LCA variant, CFP, as an environmental indicator for the system and producing information to appropriately direct the development of the novel process.

This will results in the following outputs:

a. Identification of the environmental profile and CFP of a proposed commercial scale mayonnaise-like emulsion product, manufactured using rape and sunflowerseed oil-bodies
b. Analysis of the extent to which the CFP and LCA results provide consistent data to enable correct targeting of process improvements at the early development stage
c. Indication of the potential performance of the novel system when certain key system parameters are varied.

8.1. CURRENTLY PUBLISHED DATA

A search of available literature concerning the use of oil-bodies (OB) within mayonnaise formulations was performed using techniques as described in previous chapters. Unsurprisingly given the novel nature of the proposed process, only one piece of published work was found, which was a US patent (Deckers et al., 2001). This patent, which covered the generation of novel emulsion formulations containing oil-bodies, also outlined the use of oil-bodies within a variety of emulsion-based cosmetic, pharmaceutical, industrial and food products, including mayonnaise.

8.2. PROPOSED PROCESS ROUTE

As stated in chapter 6, whilst the composition of mayonnaise varies, it has set minimums for oil content with the US Food and Drug Administration (FDA) regulation (21CFR169.140) stipulating a minimum of 65%, whilst the European Federation of Condiment Sauce Industries (FIC) recommends a minimum of 70%. The emulsion generated using oil-bodies cannot therefore be termed 'mayonnaise' at this point, but must instead be called a 'mayonnaise-like oil-body emulsion'. For simplicity within this work however, this material will be referred to throughout as 'oil-body mayonnaise' (OBM).

The only source of formulation data within literature was the Deckers et al. (2001) patent, which replaced a modest amount of the edible oil with OB material, providing a 4.98% contribution of OB. From discussions with Dr Tim Foster, Associate Professor and Reader in Food Structure within the University of Nottingham (Foster, 2012), a revised formulation was proposed that exploited the natural emulsion characteristics of the OB material. Within this formulation, the OB material was assumed to replace all of the oil and the emulsifying agent (egg) within the OBM formulation, with the remaining ingredients adjusted in line with the proposals from Decker at al. (2001). The resultant composition was as shown in table 8.2-1.

Table 8.2-1: Comparisons of conventional mayonnaise composition with that used for OBM LCA

	Conventional Mayonnaise **% input**	**Proposed WOB 'Mayonnaise-like' emulsion** **% input**
Reference	**Meeuse et al. 2000**	
Sunflower oil	80	0
Egg yolk	8	0
Water	7	1.6
Vinegar	3	9
Salt	1	0.5
Sugar	1	0
Wet oil-bodies	0	88.9
N.B: WOB = wet oil-bodies; OBM = mayonnaise-like oil-body emulsion		

As a naturally occurring emulsion, the use of OB cream to produce the OBM would not require the same multi-stage emulsification equipment as necessary for emulsification of the raw ingredients for conventional mayonnaise. Instead, it was anticipated that mixing equipment alone would be necessary to incorporate the additional ingredients to the OB cream to produce the OBM, representing a simplified production route compared with current mayonnaise manufacture.

8.3. **SYSTEM DEFINITION**

The use of the OB material within a mayonnaise formulation was at this stage a theoretical concept and as such no process data was available, even at the laboratory stage. The same approach was therefore adopted as utilised during the examination of the OB production process, with the LCA models detailed in chapter 7 used as the principal ingredient.

The product system analysed was the production of a mayonnaise-like oil-body emulsion using OB extracted from rape and sunflowerseeds. The functional unit (FU) used for the analysis was '1 tonne of rapeseed / sunflowerseed mayonnaise-like oil-body emulsion produced in UK, packaged in 600g jars, palletised and ready for distribution'. As a cradle to gate study, the starting boundary was the extraction of raw materials, which translated to cultivation of crop and rearing of animals for the agricultural products involved. The finishing boundary was the exit of the product packaging facility, thereby excluding use and disposal stages of the life cycle. The system flow digram is shown in figure 8.3-1.

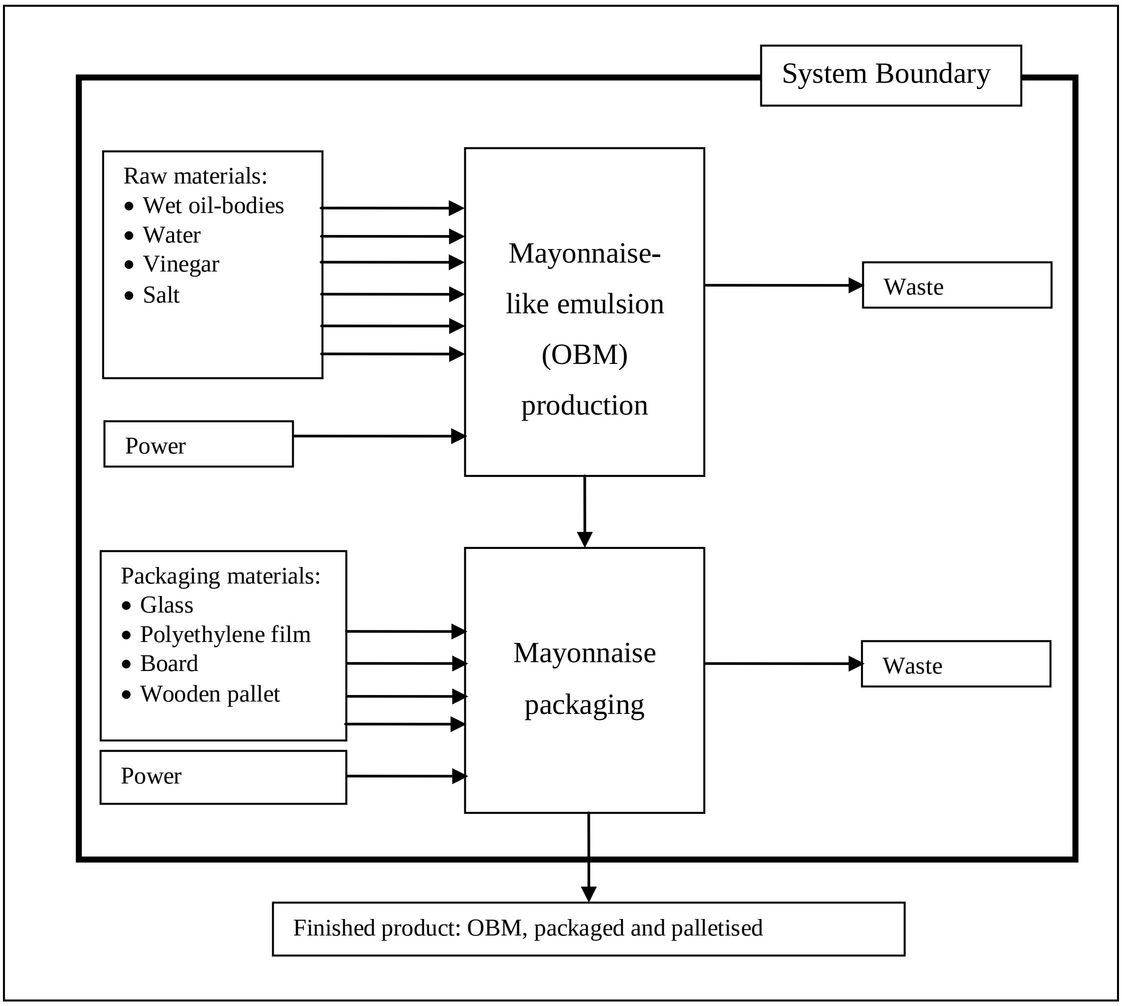

Figure 8.3-1: System flow diagram for mayonnaise-like oil-body emulsion LCA

As for the conventional mayonnaise system, attributional LCA models were constructed within SimaPro 7.3.2 for both OBM systems incorporating rapeseed and sunflowerseed oil using the OBM composition as stated in table 8.2-1. The WOB models discussed in chapter 7 were utilised as raw material within these models and Life Cycle Impact Assessment (LCIA) was again conducted using the ReCiPe(2008) method.

The wet oil-body (WOB) models generated in chapter 7 incorporated no allocation of system burdens, such that all of the impacts from the system were attributed to the WOB product, rather than any by-product. This was different to the way that the seed oil systems were treated within the conventional mayonnaise model. Within these systems, the impacts identified were attributed to both the seed oil product and the co-products generated within the extraction and refining stages, both of which could be used as animal feed. This had the combined effect that only 71.6% of the burdens within the seed oil system were attributed to rapeseed oil and 73.7% were attributed to the sunflowerseed oil.

To ensure an equitable comparison, the analyses of the mayonnaise-like oil-body emulsion systems were initially conducted by comparing the results of the unallocated OBM with those generated for conventional mayonnaise without any allocation in the seed oil process systems. The resultant models represented the environmental burdens of the entire product systems including any material that could be used for co-products.

It is anticipated that the residue from the WOB production will also be useful as a saleable product, with potential routes currently identified as either animal feed or as feedstock for bio-fuel production. As such, the principle of allocation is as appropriate for the WOB system as it is for the conventional processing route. Therefore the OBM was re-assessed using a set of different WOB models within which the calculated impacts were attributed to both the oil-body product and the co-product stream.

Economic allocation was used to ensure consistency with previous modelling, however since both product and by-product were materials that had previously not had a sale price attributed to them, assumptions needed to be made that the sale price of the WOB material would match that of the raw oil and the price of the residue stream would match those of the meal. In addition, the process was further assessed using mass allocation to assessthe impact of using alternative allocation methods, in line with the requirements of ISO 14040:2006.

8.4. DATA GATHERING

There were no new data requirements for the OBM system since all process inputs had already been utilised within previous models, however as outlined in chapter 6, the power consumption obtained for the mayonnaise production facility was not disaggregated in any way and therefore covered the emulsification and packaging processes.

Since the power requirements for emulsification were previously indicated as negligible relative to the other power inputs to the system (Dumelin (pers.comm), 2010; McKeown (pers.comm), 2011), no adjustment of this amount was made for the reduced power required for mixing rather than emulsification.

8.4.1. Determination of yield

From the data presented in table 7.5-1 it was possible to determine the WOB yield from seed, by using equation 8.4-1. Through the use of 4615 kg of seed to produce 1000 kg of WOB, the WOB yield in each case was evaluated as 21.7% .

$$\text{WOB yield} = \frac{\text{Mass of WOB produced}}{\text{Mass of seed}} \times 100 \qquad \textbf{Eq 8.4-1}$$

However within oil-body literature, the term 'yield' is often used to describe the dry-basis yield of material, which refers to the yield of oil within the oil-body material compared with that theoretically available from the seed. It was therefore prudent to determine yield on this basis also.

The WOB material has a water content of approximately 40% (Khosla, 2010) which is derived from the extraction medium. The oil-bodies (OB) therefore comprise 60% of the WOB mass, from which the dry basis yield can be calculated using equation 8.4-2.

$$\text{Dry basis yield} = \frac{\text{Mass of OB produced x lipid content}}{\text{Mass of seed x oil content}} \times 100 \qquad \textbf{Eq 8.4-2}$$

The lipid (oil) content of the OB material was approximated as 85%, based on data supplied by Khosla (2012) and as discussed in chapter 2, an accepted estimate of the oil content of the seeds under investigation was 40%, although in reality the sunflower oil content varies considerably from this. The 21.7% WOB yield previously calculated therefore translates into a 27.6% dry basis yield, as shown through equation 8.4-3, with dry basis yield equating to 1.275 times the value for WOB yield as shown in equation 8.4-4.

$$\text{Dry basis yield} = \frac{1000 \times 0.6 \times 0.85}{4615.2 \times 0.4} \times 100 = 27.6\% \qquad \textbf{Eq 8.4-3}$$

$$\text{Dry basis yield} = \text{WOB Yield} \times \frac{0.6 \times 0.85}{0.4} = 1.275 \times \text{WOB Yield} \qquad \textbf{Eq 8.4-4}$$

8.5. RESULTS AND DISCUSSION

Using models as outlined in section 8.3, analysis was performed to extract the following information, to support research objectives 4, 5 and 6.

1.	The environmental loads for the production of rape and sunflowerseed OBMs using three different allocation scenarios	**Objective 4**
2.	Comparison of these loads with those from conventional mayonnaise production	**Objective 4**
3.	The relative contributions of each of the individual OBM processing stages to the most prominent impact categories using three different allocation scenarios	**Objective 4**
4.	The extent to which CFP and LCA results provide consistent data to enable correct targeting of process optimisation efforts	**Objective 5**
5.	The WOB yield of OB material required from the seed for the novel process to match the performance of conventional processing within the key impact areas	**Objective 6**
6.	The impact of varying the process elements identified as main contributors to the key environmental impact areas.	**Objective 6**

8.5.1. Environmental loads of the OBM production – no allocation.

The unallocated characterised impact results for OBM and mayonnaise produced using both types of seed are shown in table 8.5-1. From these results it is evident that when comparing unallocated data, the environmental impacts generated through the use of WOB rather than seed oils for the production of the emulsion were considerably higher in each impact category. This is predominently due to the increased burdens associated with seed cultivation within the OBM systems, which require 4.1 tonnes of seed compared with the 2.1 tonnes required within the unallocated conventional mayonnaise systems.

Table 8.5-1: Characterised midpoint data for unallocated OBM and mayonnaise systems

Impact category	Unit	Rapeseed mayonnaise *No allocation*	Rapeseed OBM. *No allocation*	Sunflowerseed mayonnaise. *No allocation*	Sunflowerseed OBM. *No allocation*
Climate change	kg CO_2 eq	3331	5693	3646	6118
Ozone depletion	kg CFC-11 eq	0.0003	0.0005	0.0003	0.0004
Human toxicity	kg 1,4-DB eq	555	1000	405	685
Photochemical oxidant formation	kg NMVOC	12.0	18.9	14.2	22.9
Particulate matter formation	kg PM_{10} eq	7.6	13.0	5.6	9.1
Ionising radiation	kg U_{235} eq	296	503	339	571
Terrestrial acidification	kg SO_2 eq	37.7	62.9	19.1	26.8
Freshwater eutrophication	kg P eq	0.9	1.7	0.9	1.7
Marine eutrophication	kg N eq	31.4	57.7	95.0	182
Terrestrial ecotoxicity	kg 1,4-DB eq	129	253	12	23.3
Freshwater ecotoxicity	kg 1,4-DB eq	37.3	72.3	46.3	89.5
Marine ecotoxicity	kg 1,4-DB eq	11.1	20.5	10.4	18.8
Agricultural land occupation	m^2a	7106	13410	16865	32525
Urban land occupation	m^2a	69.3	133	111	212
Natural land transformation	m^2 a	0.6	0.9	0.6	0.9
Water depletion	m^3	19.2	41.8	17.8	38.3
Metal depletion	kg Fe eq	125	252	123	239
Fossil depletion	kg oil eq	724	1129	730	1069
N.B. The top five impacts as identified through endpoint normalisation are indicated by shading					

From the results presented in chapters 6 and 7, the most prominent impact categories as identified at endpoint normalisation had been determined as agricultural land occupation (ALO), fossil depletion (FD) and climate change (CC) for both sunflower and rapeseed systems, with terrestrial eco-toxicity (TET) indicated as fourth within the rapeseed system and particulate matter formation (PMF) indicated as fourth within the sunflowerseed systems. These have been highlighted by shading within table 8.5-1 and were further identified as the most prominent impact categories within the OBM system as illustrated in figure 8.5-1.

The graphs showing the full range of normalised impacts for both OBM systems are available in figures F8.1 and F8.2 in Appendix F.

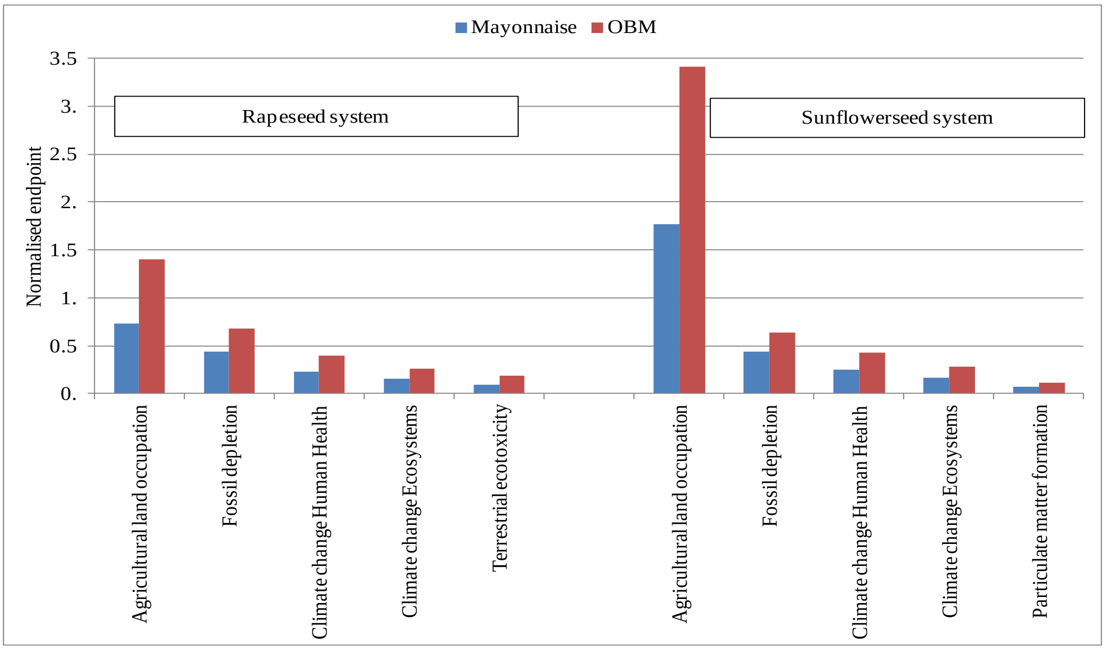

Figure 8.5-1: Top five normalised endpoint results for OBM and mayonnaise systems: No allocation

As with the results presented in previous chapters, these figures were driven by the over-riding contribution from the cultivation of the seed, with the prominently higher values for ALO within the sunflowerseed system being a result of the reduced yields for sunflower cultivation.

Full breakdowns of the contributing processes within each impact category are provided in section 8.5.4.

8.5.2. **Comparison of LCIA using economic allocation**

The characterised midpoint results for the OBM and mayonnaise models using economic allocation are shown in table 8.5-2 within which the most prominent impact categories have again been indicated by shading.

Table 8.5-2: Characterised midpoint data for OBM and mayonnaise systems with economic allocation

Impact category	Unit	Rapeseed mayonnaise *Economic allocation*	Rapeseed OBM. *Economic allocation*	Sunflowerseed mayonnaise. *Economic allocation*	Sunflowerseed OBM. *Economic allocation*
Climate change	kg CO_2 eq	2694	3364	2955	3734
Ozone depletion	kg CFC-11 eq	0.0003	0.0003	0.0002	0.0003
Human toxicity	kg 1,4-DB eq	452	604	347	451
Photochemical oxidant formation	kg NMVOC	9.6	11.2	11.4	13.9
Particulate matter formation	kg PM_{10} eq	6.0	7.4	4.7	5.6
Ionising radiation	kg U_{235} eq	245	305	279	354
Terrestrial acidification	kg SO_2 eq	29.5	35.1	16.4	16.6
Freshwater eutrophication	kg P eq	0.7	1.0	0.7	1.0
Marine eutrophication	kg N eq	23.1	30.6	70.6	101
Terrestrial ecotoxicity	kg 1,4-DB eq	92.5	134	8.9	13.0
Freshwater ecotoxicity	kg 1,4-DB eq	27.6	39.4	34.9	50.8
Marine ecotoxicity	kg 1,4-DB eq	8.9	12.1	8.5	11.6
Agricultural land occupation	m^2a	5210	7219	12539	18202
Urban land occupation	m^2a	51.9	73.2	83.5	120.4
Natural land transformation	m^2a	0.5	0.6	0.5	0.6
Water depletion	m^3	17.5	24.2	16.5	23.3
Metal depletion	kg Fe eq	96.7	142	96.8	141
Fossil depletion	kg oil eq	624	733	630	722

As with the unallocated system, it was evident that the environmental burdens for production of the OBM were prominently higher than those attributed to the production of conventional mayonnaise. However, the use of allocation prominently reduced the magnitude of the difference between the rapeseed OBM and conventional mayonnaise production. Whilst originally identified as 70% higher than conventional production, the rapeseed OBM system was calculated as having a CFP of only 24.9% higher than the conventional mayonnaise when economic allocation was used. Likewise the sunflowerseed OBM had a CFP that was 26.4% higher than that its conventional counterpart, a prominent reduction from the 67% difference determined in the unallocated system.

The normalised endpoint values for both systems are shown in figure 8.5-2. From these it was evident that whist the magnitude of the normalised impacts were reduced through attributing only a portion of the system burdens to the OB product based on the value of that product stream, the most prominent impact categories remained consistent as ALO, FD, CC for both systems. These were again followed by TET for the rapeseed system and PMF for the sunflowerseed system. Within these impact categories it was evident that the OBM systems had prominently higher environmental loads, with the full results shown in figures F8.3 and F8.4 with appendix F showing this to be the case within every impact category.

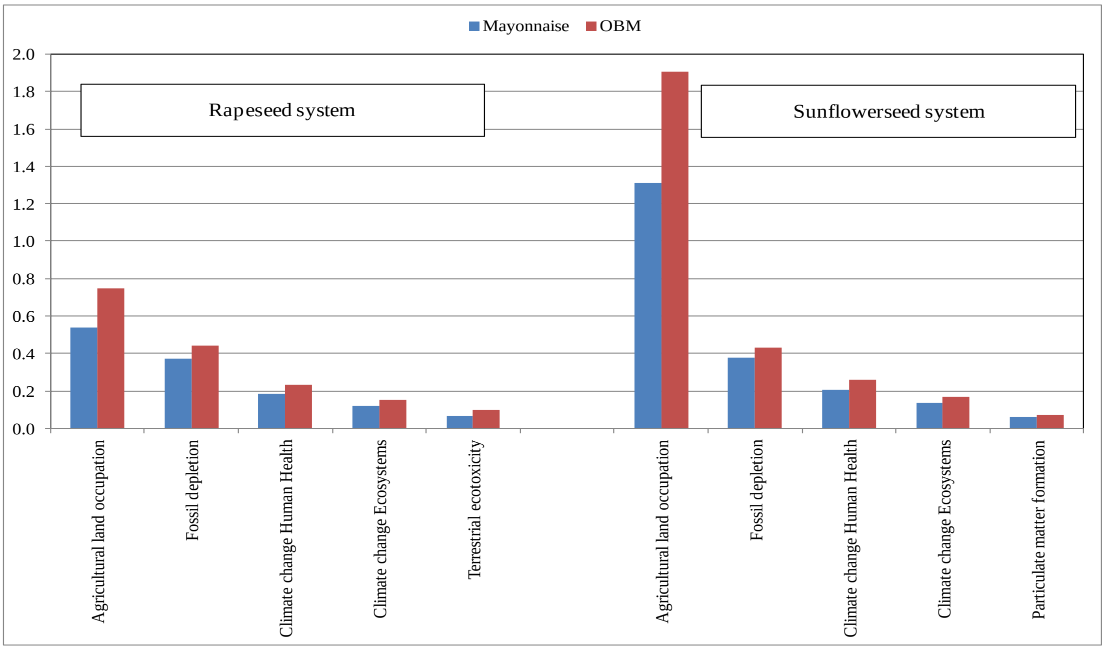

Figure 8.5-2: Top five normalised endpoint results for OBM and mayonnaise systems – economic allocation

This is predominantly due to the continued prominence of the seed cultivation within the system; the relative contributions from each process stage will be presented in section 8.6.4.

8.5.3. **Comparison of LCIA using mass allocation**

The characterised midpoint results for the OBM and mayonnaise systems analysed using mass allocation are shown in table 8.5-3 with the most prominent impact categories again indicated by shading.

In this instance the sunflowerseed OBM system had lower levels of environmental impacts than its conventional counterpart within nine of the eighteen impact categories, as shown in figure 8.5-3, with the rapeseed system exhibiting lower levels of impacts in ten of the eighteen midpoints, as shown in figure F8.5 in appendix F.

Table 8.5-3: Characterised midpoint data for OBM and mayonnaise systems with mass allocation

Impact category	Unit	Rapeseed mayonnaise *Mass allocation*	Rapeseed OBM. *Mass allocation*	Sunflowerseed mayonnaise. *Mass allocation*	Sunflowerseed OBM. *Mass allocation*
Climate change	kg CO_2 eq	1949	1821	2033	1913
Ozone depletion	kg CFC-11 eq	0.0002	0.0002	0.0002	0.0002
Human toxicity	kg 1,4-DB eq	331	341	270	273
Photochemical oxidant formation	kg NMVOC	6.8	6.1	7.6	7.0
Particulate matter formation	kg PM_{10} eq	4.2	3.8	3.4	2.9
Ionising radiation	kg U_{235} eq	185	174	198	189
Terrestrial acidification	kg SO_2 eq	20.0	16.6	12.9	8.8
Freshwater eutrophication	kg P eq	0.5	0.5	0.5	0.5
Marine eutrophication	kg N eq	13.4	12.6	37.9	39.6
Terrestrial ecotoxicity	kg 1,4-DB eq	49.8	54.9	4.7	5.2
Freshwater ecotoxicity	kg 1,4-DB eq	16.3	17.6	19.7	21.3
Marine ecotoxicity	kg 1,4-DB eq	6.3	6.5	6.0	6.1
Agricultural land occupation	m^2a	2991	3118	6754	7260
Urban land occupation	m^2a	31.5	33.6	46.9	50.6
Natural land transformation	m^2a	0.4	0.4	0.4	0.4
Water depletion	m^3	15.5	12.5	14.9	11.8
Metal depletion	kg Fe eq	63.5	69.2	60.9	66.3
Fossil depletion	kg oil eq	507	470	496	457

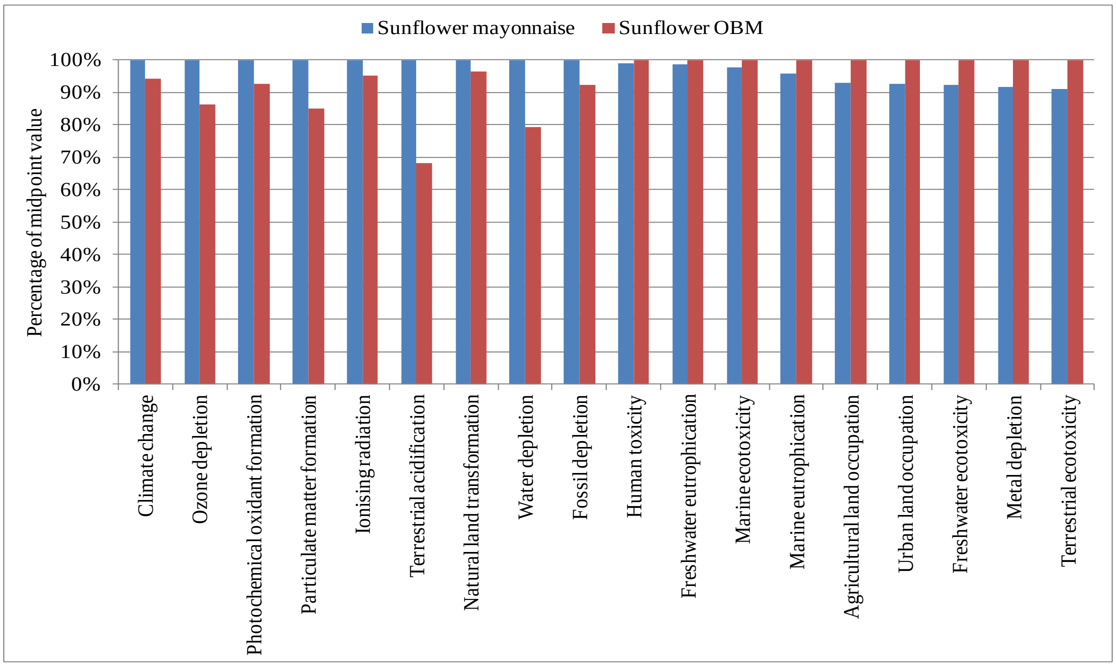

Figure 8.5-3: Percentage of characterised midpoint values - sunflowerseed systems

This improved environmental profile was borne out of the reduced proportion of the inventory that was attributed to the OB product within the OB production process, within which the reported WOB yield of 21.7% resulted in only 21.7% of the WOB process burdens being attributed to the OB product. The prominent reduction over the 52.9% and 55.6% that was attributed to rape and sunflower OBs using economic allocation led to large reductions in the OBM burdens. In this instance the use of mass allocation appears to make a low WOB yield appear advantageous and so could appear to be counter- productive to any aims to reduce the environmental burdens as a whole.

The impact categories where the OBM systems had higher levels than those from conventional mayonnaise production were those which were borne almost exclusively from the cultivation stage, namely human toxicity, marine eutrophication, terrestrial eco-toxicity, freshwater eco-toxicity, marine eco-toxicity, agricultural land occupation and urban land occupation. This indicated that even with the reduced burdens attributed through mass allocation, cultivation of the seed provided the over-riding impacts. This result was consistent with the endpoint analysis, as demonstrated in figure 8.5-4, which provides normalised results for the largest five impacts, with full profiles provided in figures F8.6 and F8.7 in appendix F.

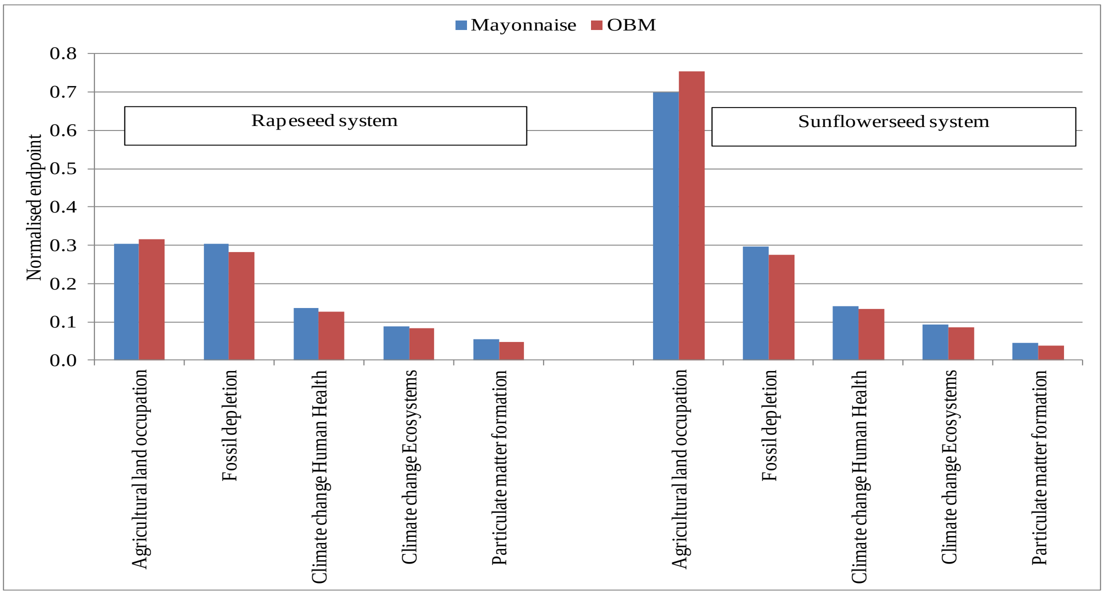

Figure 8.5-4: Comparison of conventional rapeseed mayonnaise with OBM – mass allocation

Use of mass allocation slightly changes the order of significance of the impacts as ranked using endpoint normalisation, with TET being relegated to sixth for the rapeseed OBM and PMF featuring as fifth in both systems. As such, whilst analysis of the system using mass allocation still identifies the OBM has having higher levels of TET impact than conventional mayonnaise, this no longer features as one of the five most prominent, due to the reduced impacts from cultivation through allocation by mass.

Of the five most prominent midpoints, as indicated by endpoint normalisation, mass allocation attributes the OBM impacts as lower in all categories except ALO, which ranks as the most prominent. Thus, if the performance of the system were to be assessed based on a single impact category alone for example the CC value, which provides the CFP, the system would appear entirely beneficial whereas viewing the wider range of environmental impacts would paint a different picture.

Whilst allocation by a physical parameter such as mass is indicated as the most preferable allocation method (after system expansion) in ISO 14040: 2006, the analysis conducted here demonstrates clearly that for this system, results obtained using mass allocation should be treated with great care.

8.5.4. **Relative contributions within each system**

In order to ascertain where process improvements need to be targeted to enable the novel process to match the environmental performance of the conventional technology, it was important to understand which elements of the process yielded the highest contributions to the most prominent impact categories.

Having consistently identified the same impact categories as most prominent through endpoint normalisation, each of the system models was analysed to ascertain the relative contributions of each process element to those midpoint impact categories. For both seed oil systems, agricultural land occupation, fossil depletion and climate change had previously been identified as the top three, with particulate matter formation as the fourth for the sunflowerseed oil system and terrestrial eco-toxicity the fourth for the rapeseed system.

Identification of the key process contributors not only enabled the determination of the process elements most effective to target for environmental improvements in each impact category, but also provided the evaluation of the extent to which analysis of the system using CFP alone would correctly target the development of the process. This was particularly important given the observation in section 8.5.3 that the use of CFP as a single measure of environmental performance for the OBM system modelled using mass allocation would give the impression that the OBM process had improved environmental credentials compared with the conventional process.

Through viewing the relative process contributions to the climate change midpoint value at each of the allocation scenarios, the top five contributors were indicated as seed cultivation, road transportation of seed from the farm, glass for the mayonnaise jar, gas used as fuel at the food processor and the sodium bicarbonate used for soaking the seed. These are indicated in figure 8.5-5, the detailed data for which can be viewed in tables F8.1 to F8.6 in appendix F.

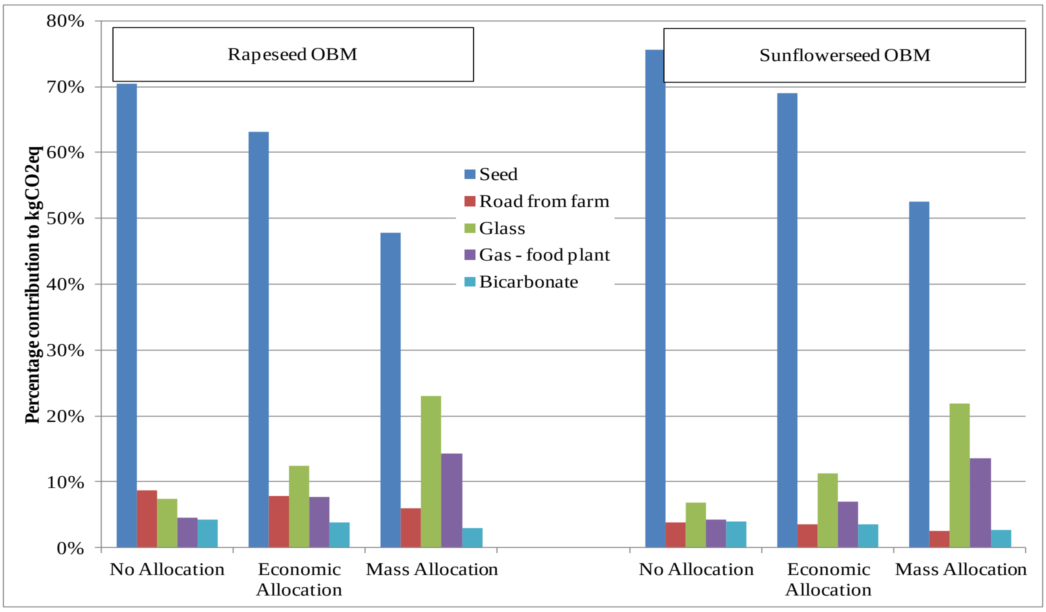

Figure 8.5-5: Percentage contributions from top five process contributors to climate change midpoint (CFP)

Irrespective of allocation method used, the largest reduction in CFP would arise if the contribution from seed could be further reduced. Whilst optimising the cultivation process is beyond the scope of the research presented here, process improvements could be targeted at maximising the OB yield from the seed, such that less seed was required. This would not only reduce the impact from the cultivation stage, but also from the transportation of seed and the quantity of sodium bicarbonate required for pre-soaking, thereby having the scope for considerable CFP reductions. Likewise, if the pre-soaking of the seed could be performed without the need for sodium bicarbonate, this would also remove the impacts from the fifth largest system contributor.

In section 8.5.3, it was shown that modelling the system using mass allocation provided results indicating the OBM process as advantageous in approximately half of the impact categories. In addition, for the most prominent impact category, ALO further improvement would be required for the OBM system to have lower ALO results than conventional mayonnaise production. For this impact area, in addition to TET, which ranked as fifth for the rapeseed OBM when viewing unallocated or economically allocated data, the over-riding contributions came from seed cultivation. Targeting process improvements on improved OB yield from seed would therefore not only reduce the CFP, but have beneficial effects across the entire range of impact categories since the environmental burdens from seed cultivation were highest within every impact category assessed.

Detailed data for all prominent impact categories is provided within tables F8.1 to F8.6 in appendix F, however for the ALO and TET results, the contribution from seed ranged from 97.8% and 99.8% for the rape and sunflower OBM systems assessed using no allocation to 91.2% and 99.7% for the OBM systems assessed using mass allocation. With such dominant seed contributions, it is evident that the improvement priorities identified through reviewing the CFP data would also greatly improve the ALO and TET credentials.

Of the five most prominent impact categories, the one with the least contribution from seed cultivation was fossil depletion, for which the relative contributions from the top five process stages is shown in figure 8.5-6. Although having less prominence within the system, the levels of FD impacts derived from cultivation still caused the seed to be the highest contributor in all LCIAs apart from that for the sunflower OBM modelled using mass allocation.

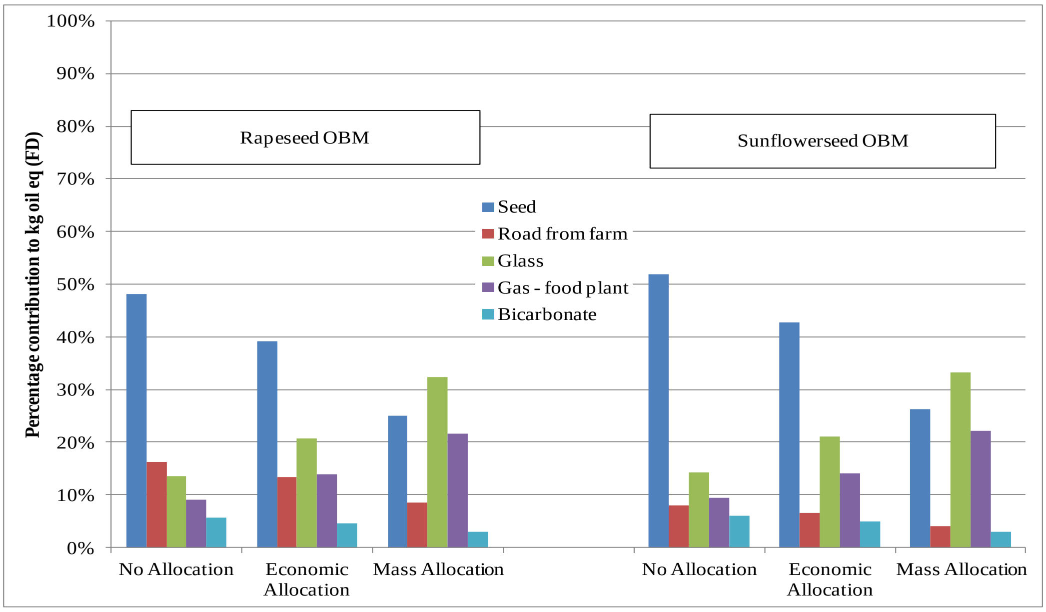

Figure 8.5-6: Percentage contributions from top five process contributors to fossil depletion midpoint

Despite this, the top five process contributors were consistent with those identified for the other prominent impacts irrespective of allocation method. As such it can be concluded that as indicated in section 8.6.3, whilst the use of a single impact category as an indicator of environmental performance could be misleading when viewing the system using mass allocation, each of the most prominent impact categories highlight the same elements of the process as the largest contributors of environmental burdens. Therefore, using results from any of the most prominent impact categories to direct the development of the novel process would lead to improvements throughout the environmental profile.

Further impact reductions from seed cultivation could be investigated through reducing or removing the need for drying the seed after harvesting. Since the OB production route involves aqueous processing, it has been suggested (Gray, 2012) that seed with a higher moisture content could be used within the process. Whilst the feasibility of such a modification could be limited by microbial growth issues during storage or transport, the 'wetter' harvesting could be coupled with more localised processing to take advantage of the greatly simplified processing equipment and procedure involved. This would further reduce the impacts from the transportation element of the system.

The environmental potential of such adaptations will be further investigated in section 8.6.

8.6. ENVIRONMENTAL IMPACT OF PROCESS IMPROVEMENTS

As with all novel processes, a considerable amount of work could be performed to optimise the performance of the WOB production and in so doing, further improve the environmental credentials of the OBM system. Having determined which of the processing elements would be most beneficial to target for optimisation, an assessment was performed of the effect that changes in the key process elements would have on the system burdens within the system using economic allocation. The process elements chosen for investigation were those that could be influenced by changes in the WOB process, as outlined in the previous section.

8.6.1. Investigation of OB yield from seed

In its current form, the production of OBM using rape or sunflowerseed WOB material would have worse environmental credentials than the conventional mayonnaise production when assessed using economic allocation or no allocation at all. In addition, when assessed using mass allocation, whilst both types of OBM have improved credentials in approximately half of the impact areas, two of the five most prominent impact categories still indicate conventional mayonnaise to have lower environmental burdens.

As stated, a considerable amount of work could be performed to optimise the performance of the WOB production and the data presented in section 8.5.4 clearly indicated that reduction in the burdens attributed to seed within all system scenarios would be the most beneficial area to target. An analysis of the OB yield required from both seed types was therefore carried out to ascertain what OB yield would be required for each of the most prominent impact categories to have results equal to those of the conventional system.

Charts showing the results of reducing the seed required for the WOB process in each prominent impact category with all three allocation models are shown in appendix F, figures F8.8 to F8.28, with a summary of the allocated data provided in tables 8.6-1 and 8.6-2.

Table 8.6-1: Percentage WOB yield required from seed for OBM and conventional systems to be equal

	Percentage RAPESEED WOB yield required for parity*			Percentage SUNFLOWER WOB yield required for parity		
Allocation basis ➔	None	Economic	Mass	None	Economic	Mass
ALO	41.5	32.0	23.0	42.0	31.5	23.5
FD	43.0	29.0	X	39.0	28.0	X
CC	42.0	29.5	X	41.0	29.5	X
TET	42.0	31.5	24.0			
PMF			X	39.0	27.5	X
* - Yield figures to the nearest 0.5%						
'X' indicates current OBM burdens are lower than those for conventional mayonnaise						

Table 8.6-2: Percentage dry basis yield required from seed for OBM and conventional systems to be equal

	Percentage RAPESEED dry basis yield required for parity*			Percentage SUNFLOWER dry basis yield required for parity		
Allocation basis ➔	None	Economic	Mass	None	Economic	Mass
ALO	53.0	41.0	29.0	53.0	40.0	30.0
FD	55.0	37.0	X	50.0	36.0	X
CC	54.0	38.0	X	52.0	38.0	X
TET	54.0	40.0	29.0			
PMF			X	50.0	35.0	X
* - Yield figures to the nearest 0.5%						
'X' indicates current OBM burdens are lower than those for conventional mayonnaise						

The WOB yield figures required for the non-allocated system of between 39% and 43% represent the WOB yield from seed that would be required for the inventory of the entire system to have equal burdens to those from the entire conventional mayonnaise system. Since the intention is that the co-products from the OB processes will be utilised as saleable products, either as animal feed or bio-fuel feedstock, this non-allocated data provides an early indication of whether the co-product needs to replace a material that has a higher or lower environmental burden than that of the material which is replaced by the conventional co-product.

The data extracted from the model using economic allocation indicates that an OB yield of between 29% and 32% (37% - 41% dry basis yield) would be required for parity with the conventional rapeseed oil mayonnaise system. Similar results were obtained for the sunflower system, where increases in yield up to 27.5% to31.5% (36% - 40% dry basis) would be required for the burdens of the novel system to match those of the conventional mayonnaise processing.

Whilst such yields were not obtained during the laboratory trials conducted during this project, they are well within the range of yields obtained during larger scale trials for the extraction of hemp seed oil-bodies conducted in 2008 as part of the SIPOS project that preceded the SEIBI project. Records supplied by Dr D Gray indicated the dry basis yields achieved during this project were between 50% and 77%.

With mass allocation, the novel system had lower characterised midpoint scores in all of the most prominent impacts apart from ALO. For the impacts of the novel sunflowerseed OBM to equal those of the conventional system within this category, a yield of 23.5% would be required, with 23% required within the rapeseed system. Whilst not indicated as one of the top five impact categories using mass allocation, TET had been previously indicated as prominent with the other two allocation scenarios. The yield assessment was therefore also performed in this category, where it was calculated that the rapeseed OBM would need a yield of 24% required for parity of TET midpoints.

8.6.2. **Impact of removing the seed drying stage.**

Whilst the novel processing will do nothing to change the cultivation of the seed, it has been suggested (Gray, 2012) that seed with a higher moisture content than currently supplied could be used within the process. Whilst the current supply chain for grain requires the seed to be harvested and dried for storage and transportation, the potential environmental impact of removing the need for drying was examined.

Regional variations exist that make drying more of a requirement in certain locations and climatic conditions than others. Of the seed cultivation datasets used within the aggregate LCI data for rape and sunflowerseed cultivation, three of the four rape datasets contained a distinct process input for drying of the seed and one of the two sunflowerseed cultivation datasets contained such an input. From reviewing the background literature for the datasets (Jungbluth et al., 2007; Nemecek et al., 2007), it was identified that certain datasets excluded drying as their moisture content at harvest was the same as that at storage.

The Integrated Swiss sunflower production incorporated the highest drying requirements, to reduce the moisture content of sunflowerseed from 17% at harvest, to 6% in storage. Within the rapeseed datasets, where drying was included, this was to reduce the moisture content from 12% to 6%, for both the extensive and integrated Swiss production datasets, with a more modest reduction from 10% to 9% moisture content included for the Saxony Anhalt data.

Following the same rationale used for development of the initial aggregate cultivation sets, seed may be sourced from a variety of locations, some of which will require drying and others that will not. Additional aggregate datasets were therefore generated for both rape and sunflowerseeds within which the drying components, where present, were removed. These 'No-drying' aggregates were then used within the OBM models, generating a set of comparative analyses that determined the effect that removing the drying stage had on the impacts of the system. The results of this analysis can be seen in figure 8.6-1, with further analysis shown in figures F8.32 to F8.34 in appendix F. Within each of these charts, the results for ALO and TET have been excluded as the process changes had a negligible effect.

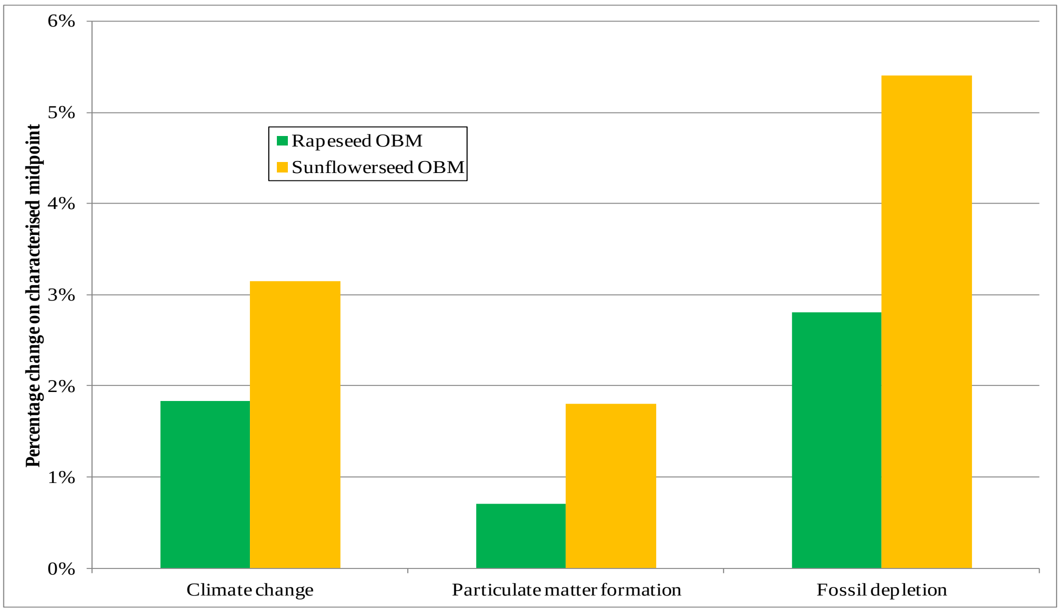

Figure 8.6-1: Percentage change in characterised midpoint through removing seed drying

For the rapeseed OBM system, the removal of the requirement for drying reduced the CFP by 62 kg CO_2eq, representing a 1.8% reduction in the rapeseed OBM. The removal of the increased drying requirement within the sunflowerseed system caused a CFP reduction of almost double at 118 kg CO_2eq, representing a 3.1% CFP reduction for the sunflower OBM.

Much smaller impact reductions were observed within the PMF category, where the levels of PM_{10} eq were reduced by 0.05 PM_{10}eq for the rapeseed OBM and 0.1 PM_{10}eq for the sunflowerseed OBM, representing 0.7% and 1.8% reductions respectively.

The largest impact reductions through removing the drying requirement were identified in the FD category, where the 21 kg oil eq represented a 2.8% reduction in FD impact for the rapeseed OBM. The results for the sunflowerseed OBM were roughly double those of the rapeseed system, with the FD midpoint decrease of 40 kg oileq generating a 4.9% reduction.

Whilst all these figures are modest reductions to the overall impacts, they do indicate that small improvements in the environmental profile could prevail if the need for seed drying was reduced.

8.6.3. **Impact of processing OB material at the farm**

The supply chain currently proposed for the WOB feedstock for the OBM process involves transportation of the seed from the farm to a processor in the same way that seed is currently transported from farms to oil-processors. This is necessary with the conventional technology since the equipment required for milling and solvent extraction makes large scale processing unfeasible at a local level.

With the simplified process proposed for the aqueous extraction of OB however, localised smaller scale processing could be possible and this would lead to reductions in the transportation element of the process since the smaller tonnages of OB material, rather than seed would transported over the longer distances.

Table B8.3 in appendix B provides details of the modifications made to the transportation elements within the LCA models to fully assess the impact of this change to the supply chain, with figure 8.6-2 showing the results obtained.

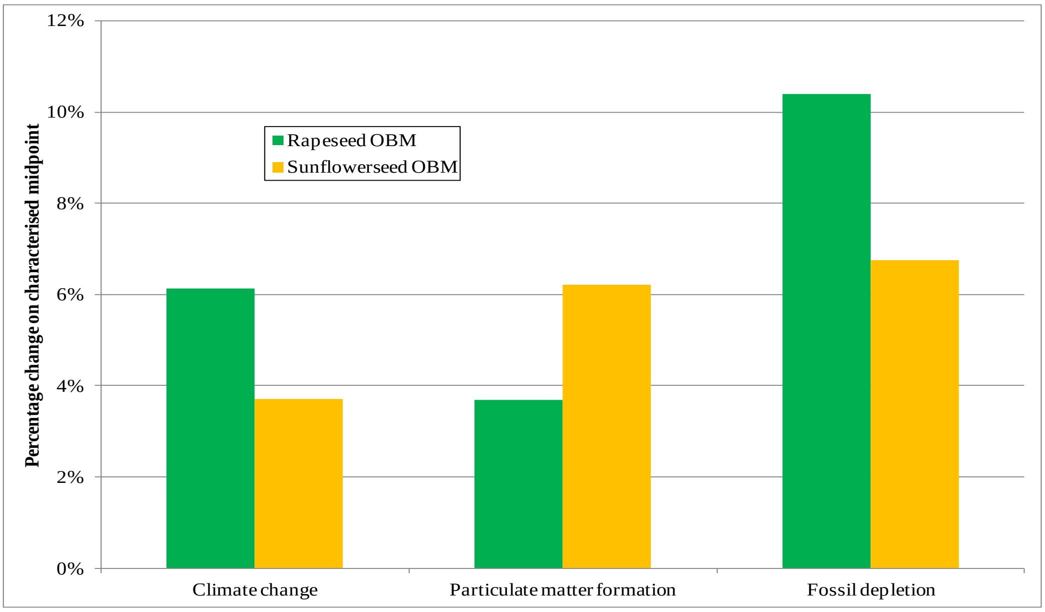

Figure 8.6-2: Percentage change in characterised midpoint through processing OB material at farm

The higher dependence on road transport within the rapeseed system had previously been highlighted as a prominent contributor within several of the impact categories, particularly climate change and fossil depletion. This is further evident through this analysis, where the reduced road transport required due to processing at the farm results in a decrease of 206 kg CO_2eq within the rapeseed OBM system, representing a 6.1% reduction in CFP, rising to a 10.4% reduction in fossil depletion impacts through the 76 kg oil eq less required due to the removal of seed transportation.

More modest changes were identified for the sunflower OBM system, due to the reduced dependence on road transport. In this instance processing at farm would lead to 135 kg CO_2eq less emissions in the CC category, giving a CFP reduction of 3.7%, with a 6.7% reduction in FD impacts.

The reductions within the PMF impact category show larger reductions than for the rapeseed OBM system, which is due to the contribution from sea transport for the seed. Further analysis of these results can again be seen in figures F8.32 to F8.34 in appendix F.

8.6.4. **Impact of removing sodium bicarbonate for seed soaking**

To aid the aqueous extraction of oil-bodies, the process currently includes a seed pre-soaking stage in which the seed is soaked in a 0.3 Molar sodium bicarbonate solution. As discussed in section 8.5.4, the bicarbonate features in the top five process contributors for each of the key impact categories and it is therefore relevant to investigate effect that removal of this stage would have. Improvement of the OB yield from seed would decrease the impacts arising from this stage by reducing the amount of seed needing to be soaked, and hence the quantity of bicarbonate required. However, the effect of removing this stage completely by soaking the seed with water only was investigated.

The sodium bicarbonate input was therefore removed from each of the system models and analysis performed to determine the levels of impact reduction achieved. The results of the 'no-bicarbonate' modelling can be seen in figure 8.6-3, with further detail within figures F8.32 to F8.34 in appendix F.

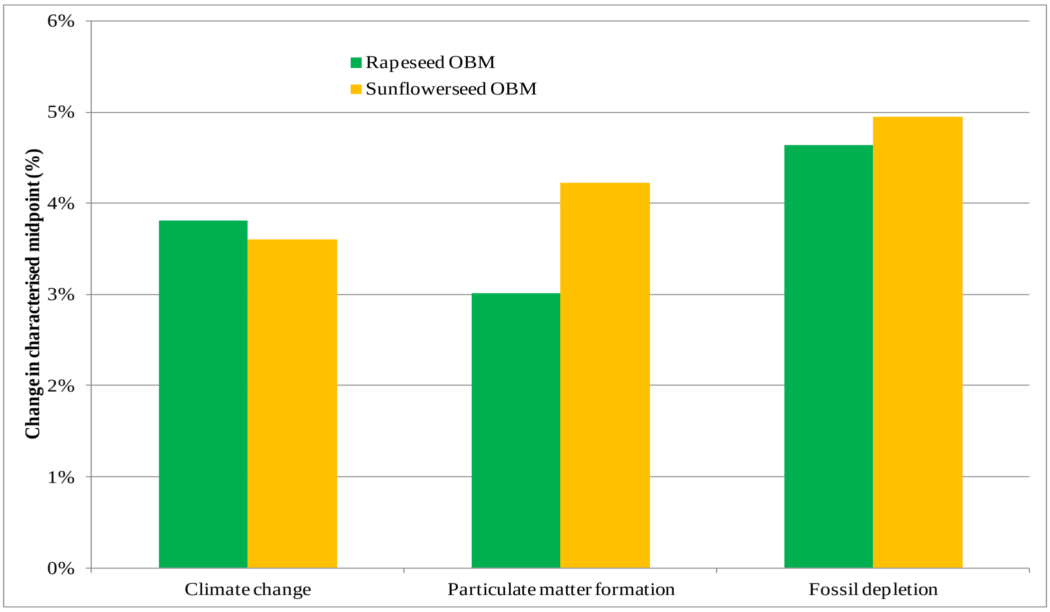

Figure 8.6-3: Percentage change in characterised midpoint through removing sodium bicarbonate pre-soaking step

The largest impact reductions for both seed types were within the fossil depletion category where the removal of the bicarbonate from the system produced reductions of 33 kgOil eq and 35 kgOil eq for the rape and sunflowerseed OBMs respectively This represented a 4.6% and 4.9% reduction in the impact value. Slightly lower improvements were calculated for the CFPs, with the 3.8% and 3.6% reductions coming from CC impact levels that decreased by 128 kg CO_2eq for the rapeseed OBM and 135 kg CO_2eq for the sunflowerseed OBM system.

An overview of these results as compared with those impact reductions obtained through the other potential process modifications can be seen in figure 8.6-4.

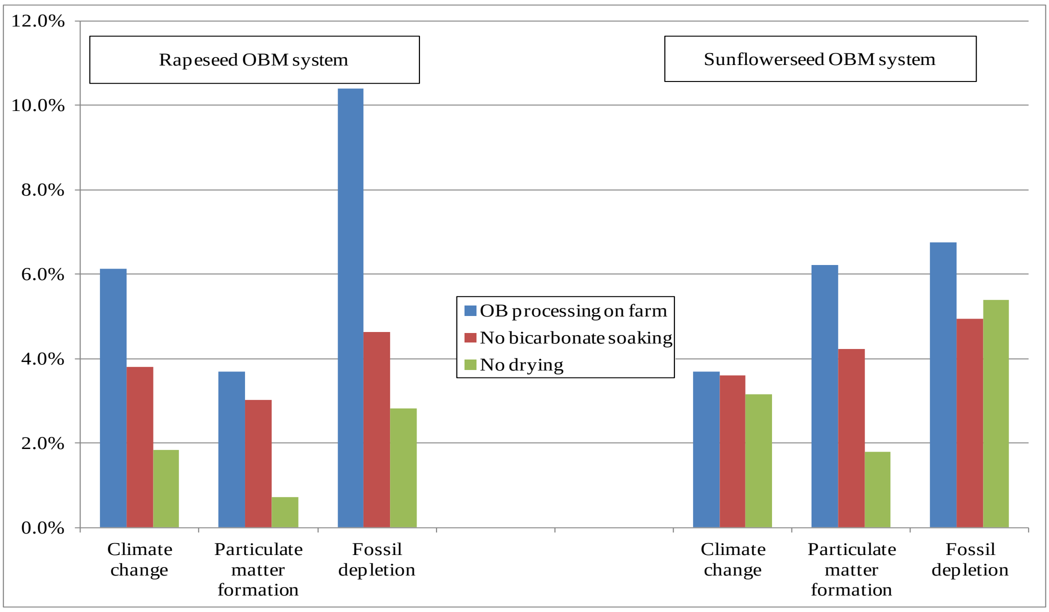

Figure 8.6-4: Overview of the effect of changing process elements on key midpoints

8.7. SUMMARY

This section will discuss the findings from the analysis of the LCA modelling performed, with particular reference to the relevant research objectives covered, which encompass the identification of environmental loads, examination of the impact of methodological choices and the generation of information to direct the ongoing development of the process.

In line with objective 4, the environmental profile and CFP for the proposed production of a commercial scale food grade emulsion using aqueously extracted rape and sunflowerseed oil-bodies has been assessed and compared against the performance of conventionally produced mayonnaise. Separate attributional LCA models were developed incorporating both economic and mass allocation, together with ones that had all seed oil allocation removed to enable the determination of the environmental burdens for the entire product system.

For each of the systems, the five most prominent impact categories were identified via endpoint normalisation. For both systems the top three were agricultural land occupation, fossil depletion and climate change, irrespective of allocation type. In the unallocated and economically allocated systems, the next most prominent category for the rapeseed OBM was terrestrial eco-toxicity, although this was relegated to sixth when allocation by mass was used, due to the reduced seed cultivation inventory levels. For mass allocation, particulate matter formation, which ranked as fifth for the sunflowerseed OBM system irrespective of allocation method, also ranked higher than terrestrial eco-toxicity for the rapeseed OBM model.

Of the five most prominent midpoints as indicated by endpoint normalisation, the models analysed using no allocation or economic allocation indicated that both OBM systems exhibited higher levels of impacts in all impact categories than their conventional mayonnaise counterparts. This differed considerably from the results using mass allocation where the OBM impacts for both systems were identified as lower in all prominent categories except agricultural land occupation, where the highest levels of impacts were indicated.

The effect of using different allocation methods and factors had been shown to be significant when determining the raw impact values for the seed oil systems in chapter 5. The analysis presented here supports these findings and further indicates that the use of mass allocation appears to make a low yield of OB from seed appear advantageous, which could be counter-productive to any aims to reduce the environmental burdens as a whole. In this instance, despite the preference within ISO 14044:2006 for allocation in a way '*that reflects the underlying physical relationships between them'*, allocation by mass appears detrimental to the use of LCA results for driving environmental improvements through process development.

ISO 14044:2006 states that '*whenever several alternative allocation procedures seem applicable, a sensitivity analysis shall be conducted to illustrate the consequences of the departure from the selected approach.'* From the analysis presented here, it could be said that wherever decisions need to be taken on the basis of attributional LCA results, sensitivity analyses should always be performed using different allocation approaches.

This is particularly the case for novel processes, where such decisions could irrevocably affect the viability of the future process. Furthermore, with the uncertainties surrounding usage of co-products for novel processes, an assessment of the burdens without allocation is also of great benefit. This not only indicates the levels of environmental impacts for the entire system, but gives an indication of the burdens that need to be offset by any saleable co-product stream.

As with all novel processes, a considerable amount of optimisation work could be performed on the OB production and in so doing, further improve the environmental credentials of the mayonnaise-like oil-body system. To identify where best to target the optimisation of the process, the relative contributions of each of the processing elements was identified for each of the most prominent impact categories. This investigation also enabled an evaluation of the extent to which analysis of the system using a single impact category such as CFP would correctly target the development of the process, in accordance with research objective 5.

As the key process contributors to CFP were consistent with those identified for the wider environmental impacts irrespective of allocation method, it was concluded that whilst the use of a single impact category as an indicator of environmental performance could be misleading when viewing the system using mass allocation, each of the most prominent impact categories highlight the same elements of the process as the largest contributors of environmental burdens. Therefore, using results from any of the most prominent impact categories to direct the development of the novel process would lead to improvements throughout the environmental profile.

On completing the analysis to compare the environmental performance of the oil-body material within a mayonnaise-like product, further analysis was performed to fulfil the sixth objective, that of producing information to appropriately direct the development of the novel process. The analysis of key process contributors had indicated that a reduction in the burdens attributed to seed within all system scenarios would be the most beneficial area to target and analysis was performed to determine WOB yield required for the OBM system to equal the performance of conventional mayonnaise in each of the top five impact areas.

With the reduced burdens identified for the mass allocated system, only agricultural land occupation had been indicated as requiring yield improvements for parity with the conventional technology, for which analysis indicated that WOB yields of 23% and 23.5% would be required for the rapeseed and sunflowerseed systems respectively.

Larger improvements were required for the system when assessed using economic allocation, with necessary WOB yield calculated as 29% to 32% (37% - 41% dry basis) dependant on category. Given the previously observed uncertainties concerning sale price for both the OB and co-product, it should be noted that the yield data obtained for economic allocation would be entirely dependent on the OB co-product being saleable for the same value as the existing oil- process meal. If it were to command a higher value, the required yield would be reduced and vice versa. This again indicates the particular importance of performing sensitivity analyses on the effect of allocation in such novel processes systems.

Whilst beyond the yield range achieved during trials within the SEIBI project, yield values in excess of these had been reported during larger scale trials for the aqueous extraction of oil-bodies from hemp seeds as part of the SIPOS project in 2008 (Gray, 2014). To estimate the potential performance of the system if such yields were to be achieved for the specific process investigated by this PhD research, analysis of the LCA at the reported yield levels of 50% and 77% (dry basis yield) was performed for the sunflowerseed system using economic allocation.

As shown in figure 8.7-1, if the OB extraction process were to attain the 50% - 77% dry basis yield levels, the environmental performance of the resultant mayonnaise-like oil-body emulsion would be considerably better than that of conventional mayonnaise in all of the most prominent impact categories, with the CFP reducing to 79% of the mayonnaise value at the 50% dry basis yield level and 61% of the mayonnaise CFP at the 77% level. This indicates the clear potential for the process.

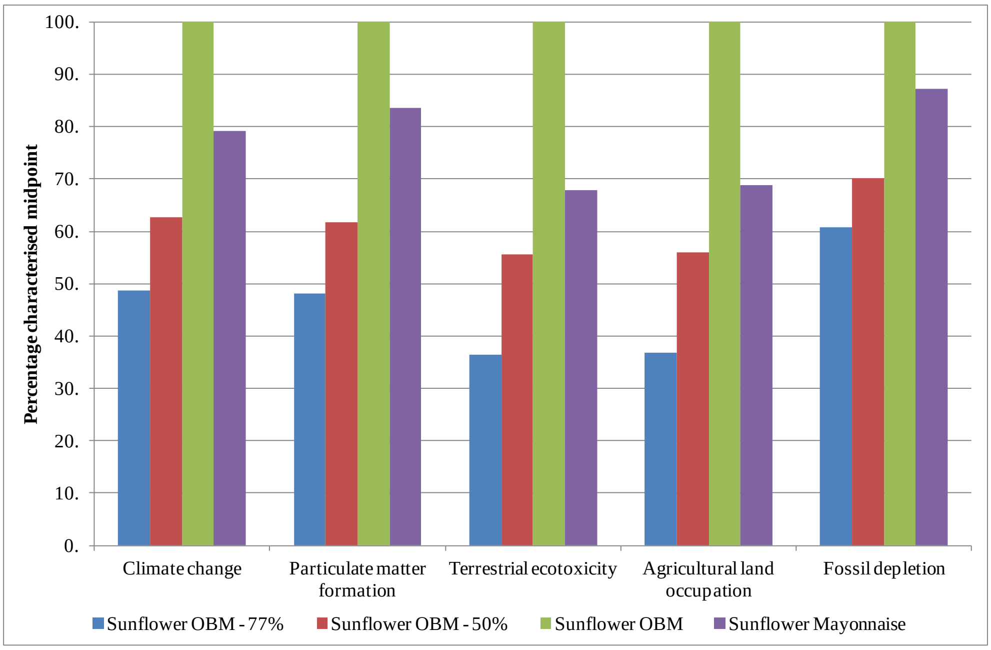

Figure 8.7-1: Percentage characterised midpoint values for mayonnaise and OBM at different dry basis yield levels

The analysis of key process contributors also indicated several other areas of potential environmental optimisation through modifications to the OBM system. The effect of removing the seed drying stage, processing the material at the farm and pre-soaking the seeds in water, rather than a sodium bicarbonate solution were all investigated using the process model developed with economic allocation. Sensitivity analysis was not performed in this instance as the results generated were merely an indication of the latent potential of individual process improvements.

Each of these modifications provided negligible reductions in the agricultural land occupation and terrestrial eco-toxicity impact values, as would be expected since these impacts derived almost exclusively from the actual cultivation of the seed. In the other prominent impact categories, modest reductions were indicated as shown in the summary table 8.7-1.

Table 8.7-1: Reductions in characterised OBM midpoint scores through process changes

	Rapeseed OBM			Sunflowerseed OBM		
	CC	PMF	FD	CC	PMF	FD
Removal of seed drying	1.8%	0.7%	2.8%	3.1%	1.8%	5.4%
Processing on farm	6.1%	3.7%	10.4%	3.7%	6.2%	6.7%
Pre-soaking without bicarbonate	3.8%	3.0%	4.6%	3.6%	4.2%	4.9%

CC = Climate change; PMF = Particulate matter formation; FD = Fossil depletion.

From this analysis, it was clearly demonstrated that the largest potential for further improving the environmental performance of the OB system, such that the OBM had equal or improved credentials over the conventional processing came from improving the OB yield from seed to provide reductions in the seed cultivation loads. The process modifications investigated all led to impact reductions, although at this stage, the modest nature of those reductions coupled with the larger potential arising from OB yield optimisation may make only those modifications that are simple to implement feasible to pursue. On that basis, the removal of the sodium bicarbonate soaking stage should be further investigated, which could easily afford impact reductions providing its removal does not lead to increased requirements elsewhere.

8.8. CONCLUSIONS

The research presented in this thesis was aimed at identifying whether the novel processing route for production of food grade edible oil emulsions via aqueous extraction of oil-bodies has a better environmental profile than that of the existing technology route. This chapter presented the culmination of the LCA modelling necessary to fulfil this aim by fully assessing the environmental credentials of the oil-body emulsion material within a functionally equivalent product system to conventional mayonnaise production.

To gain a full understanding of the impact profile of the systems, the LCA modelling was performed using both economic and mass allocation, in addition to an assessment of the system with allocation parameters removed.

The life cycle impact assessment for the unallocated system indicated that both OBM systems had prominently higher environmental burdens in every impact category, with the CFP of the rapeseed OBM being 5.19 tonnes CO_2eq, or 70% higher than the unallocated conventional mayonnaise system. Likewise the CFP for the sunflowerseed OBM was 67% higher than its conventional counterpart, at 6.12 tonnes CO_2eq.

Literature review work detailed in chapter 4 had identified economic allocation as the preferred basis for the majority of past seed oil assessments. When the system model using economic allocation was reviewed, both rape and sunflowerseed OBM systems were again identified as having higher environmental loads across the entire range of midpoint impact categories, although the allocation of burdens between OB product and co-product had reduced the CFPs to 3.36 tonnes CO_2eq for the rapeseed OBM and 3.73 tonnes CO_2eq for the sunflowerseed OBM system, which were 24.9% and 26.4% higher than conventional mayonnaise production.

Considerably different results were obtained when using mass allocation, where the low OB yield from seed led to greatly reduced burdens being attributed to the OB product stream. In this case the OBM systems were identified as having improved environmental credentials in half of the impact categories, including CC, where the CFPs were calculated as 1.82 tonnes CO_2eq for the rapeseed OBM and 1.91 tonnes CO_2eq for the sunflowerseed OBM system, compared with 1.95 tonnes CO_2eq for the rapeseed mayonnaise and 2.03 tonnes CO_2eq for the sunflowerseed mayonnaise system using mass allocation.

The dissimilarity of the results generated highlights the importance of conducting sensitivity analyses where different allocation parameters can be used. This is of particular importance in emerging processes such as investigated in this thesis where decisions taken on the basis of LCA results could irrevocably affect the viability of the future process. Furthermore, it was concluded that mass was not a suitable allocation parameter for analysis of ongoing developments with this system.

Given the popularity of CFP as an indicator of environmental performance, an assessment was performed of the extent to which the CFP and LCA results provided consistent data to enable correct targeting of process improvements. The top five process contributors to the CFP were identified as the same as for the other prominent impact categories, and it was therefore concluded that whilst the use of CFP as an indicator of environmental performance could be misleading when viewing the system using mass allocation, using results from any of the most prominent impact categories, including CFP, to direct the development of the novel process would lead to improvements throughout the entire environmental profile.

An assessment was conducted of the OB yield required from the seed for the OBM system to equal the environmental performance of the conventional mayonnaise process within the five most prominent impact categories. Agricultural land occupation was the only impact category where the mass allocated system required a yield improvement for the OBM production be equal to the conventional mayonnaise performance. In this instance a very modest yield improvement was required from the current 21.7% to 23% (29% dry basis) for the rapeseed OBM and 23.5% (30% dry basis) for the sunflowerseed OBM.

Using economic allocation, the yield required for parity between the novel and conventional systems for the rapeseed variant varied from 29% (37% dry basis) for the fossil depletion category to 32% (41% dry basis) for the agricultural land occupation. Similar yields were calculated as required for the sunflower system at 27.5% (35% dry basis) for the particulate matter formation category and 31.5 % (40% dry basis) for the agricultural land occupation.

Removal of the seed drying stage at cultivation, processing of the OB material at the farm and removal of the sodium bicarbonate from the seed pre-soaking step were all investigated to determine the level of environmental improvements that they could bring. Of these, processing at the farm had the greatest potential for savings within the prominent impact categories, with removal of the sodium bicarbonate from the pre-soaking and processing of seed with a higher moisture level both offering more modest, but still prominent savings.

Having completed the modelling and analysis of the OBM systems, comparison against conventional technology and determination of optimisation options, the work presented within this chapter has detailed the findings of the fourth, fifth and sixth research objectives. This information can now be used together with that from the preceding chapters to develop the overall conclusions from the research outlined within this thesis.

Full details of the considerations and analysis in building these conclusions will be discussed in the next chapter.

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CHAPTER 9. DISCUSSION

This chapter adds to the findings and implications of the analysis of each of the separate Life Cycle Assessments (LCAs) discussed in chapters 5 to 8 by providing an overarching discussion of areas that are common across the results, to aid with the development of the overall conclusions.

The novel output from these results could be broadly placed within two categories:

- Quantification of carbon footprint (CFP) and wider environmental loads
- Determination of the impact of methodological choices.

9.1. QUANTIFICATION OF CFP AND ENVIRONMENTAL LOADS

9.1.1. Analysis of contributing models

This part of research objectives 1 to 4, was essential for the stage-wise creation of LCA models to enable the eventual comparison of the environmental performance of the aqueously extracted oil-bodies within a functionally equivalent mayonnaise system. In addition to being building blocks for the analysis, the determination of CFP and environmental loads of each product system provided information that will be of use for researchers and LCA practitioners within the edible oils, food development and the wider LCA community. As discussed in section 3, no published data exists for either the mayonnaise or aqueous oil-body extraction process and where data has been published for rape and sunflowerseed oils as products, this has been based on the use of different methodological choices.

The five most burdensome categories when assessed using either mid or endpoint normalisation were the same for all product systems. This was principally due to the dominant contribution that cultivation of seed made within each system involved, with over 90% of the characterised values of those impacts identified as prominent through midpoint normalisation for the seed oil LCAs derived from seed. The only exception to this was natural land transformation whose impacts arose from the transformation of land through extraction of feedstock for power generation and thus attributed a higher contribution to the more power intensive process stages.

The characterised midpoint values for the impacts identified as most prominent at both mid and endpoint level are provided in table 9.1-1, with the CFP results as provided by the climate change midpoints highlighted for ease of extraction.

Table 9.1-1: Top five characterised midpoint results for seed oil systems when normalised at mid and endpoint

Characterised midpoint results Top five through MIDpoint normalisation						Characterised midpoint results Top five through ENDpoint normalisation					
RAPESEED OIL			SUNFLOWERSEED OIL			RAPESEED OIL			SUNFLOWERSEED OIL		
TET	115	kg1,4-DBeq	**ME**	86.1	kg N eq	**ALO**	6003	m^{2}a	**ALO**	15158	m^{2}a
FET	31.5	kg1,4-DBeq	**FET**	40.6	kg1,4-DBeq	**FD**	418	kg oil eq	**FD**	424	kg oil eq
ME	26.3	kg N eq	**ALO**	15158	m^{2}a	**CC (CFP)**	2271	kg CO_2 eq	**CC (CFP)**	2597	kg CO_2 eq
NLT	0.3	m^2	**NLT**	0.4	m^2	**TET**	115	kg1,4-DBeq	**PMF**	3.6	kg PM_{10} eq
FE	0.7	kg P eq	**FE**	0.8	kg P eq	**PMF**	5.3	kg PM_{10} eq	**TET**	10.9	kg1,4-DBeq
RAPESEED OIL MAYONNAISE			SUNFLOWERSEED OIL MAYONNAISE			RAPESEED OIL MAYONNAISE			SUNFLOWERSEED OIL MAYONNAISE		
TET	92.5	kg1,4-DBeq	**ME**	70.6	kg N eq	**ALO**	5210	m^{2}a	**ALO**	12539	m^{2}a
NLT	0.5	m^2	**FET**	34.9	kg1,4-DBeq	**FD**	624	kg oil eq	**FD**	630	kg oil eq
FET	27.6	kg1,4-DBeq	**NLT**	0.5	m^2	**CC (CFP)**	2694	kg CO_2 eq	**CC (CFP)**	2955	kg CO_2 eq
ME	23.1	kg N eq	**ALO**	12539	m^{2}a	**TET**	92.5	kg1,4-DBeq	**PMF**	4.7	kg PM_{10} eq
FE	0.7	kg P eq	**FE**	0.7	kg P eq	**PMF**	6.0	kg PM_{10} eq	**TET**	8.9	kg1,4-DBeq
RAPESEED WET OIL-BODIES			SUNFLOWERSEED WET OIL-BODIES			RAPESEED WET OIL-BODIES			SUNFLOWERSEED WET OIL-BODIES		
TET	284	kg1,4-DBeq	**ME**	205	kg N eq	**ALO**	14782	m^{2}a	**ALO**	36289	m^{2}a
FET	78.8	kg1,4-DBeq	**FET**	98.1	kg1,4-DBeq	**FD**	960	kg oil eq	**FD**	892	kg oil eq
ME	65.0	kg N eq	**ALO**	36289	m^{2}a	**CC (CFP)**	5598	kg CO_2 eq	**CC (CFP)**	6076	kg CO_2 eq
NLT	0.8	m^2	**NLT**	0.8	m^2	**TET**	284	kg1,4-DBeq	**PMF**	8.9	kg PM_{10} eq
FE	1.8	kg P eq	**FE**	1.8	kg P eq	**PMF**	13.3	kg PM_{10} eq	**TET**	26.1	kg1,4-DBeq

TET = Terrestrial eco-toxicity; FET = Freshwater eco-toxicity; ME = Marine eutrophication; NLT = Natural land transformation; FE = Freshwater eutrophication; ALO = agricultural land occupation; FD = Fossil depletion; CC = Climate change; PMF = Particulate matter formation.

At the midpoint level, both marine and freshwater eutrophication, freshwater eco-toxicity and natural land transformation were indicated within the top five for all systems. Terrestrial eco-toxicity was also included for the rapeseed systems, ranked as highest burden. Here, the contributions arising from pyrethroid pesticide use during cultivation led to the far higher impacts in that category for the rapeseed system. With the exception of terrestrial eco-toxicity, all of the impacts derived from cultivation were higher within the sunflower system due to the reduced yields, as supported by Shonfield and Dumelin (2005) and Iriarte et al. (2010). This was also shown through the increased significance of the agricultural land occupation impacts, which rose to third most prominent in all sunflower product systems.

As outlined in chapter 3, midpoints rely primarily on scientific information and well proven facts, leading to a lower level of uncertainty than with endpoints (Goedkoop et al. 2013). Non-toxic impact categories such as climate change or acidification are assumed to be relatively accurate through the use of comprehensive characterisation factors, based on well reported emission data (Laurent et al. 2012). Greater levels of uncertainty arise however when assessing toxic impacts, or those categories that involve large numbers of chemicals, since both emission inventories and characterization factor databases are unable to cover the thousands of chemicals released to the environment during the life cycles involved (ibid).

Since the midpoints identified as most prominent through midpoint normalisation were predominantly those with the highest levels of uncertainty, the approach advocated by Van Hoof (2013) was adopted, using endpoint normalisation to identify the most prominent impact categories, but viewing the characterised midpoint data within these categories.

Given the consistency of impact ranking and the uncertainty surrounding results from midpoint normalisation, only endpoint normalisation was used for the final system analysis involving the mayonnaise-like oil-body emulsion.

9.1.2. Mayonnaise-like oil-body emulsion (OBM) environmental loads

The results generated for the oil-body extraction process in chapter 7 and presented in table 9.1-1 were generated with a model that did not contain any allocation of co-products. These values represented the environmental burdens for the entire product system, including any saleable co-products. The residues generated through the aqueous extraction process were proposed to be used as a product stream, for sale as either animal feed or bio-fuel feedstock. As such, whilst knowledge of the entire system burdens would be useful from a 'whole system' development perspective, the treatment of co-product streams using allocation was as appropriate for the novel system as it had been for the conventional processes.

ISO14044:2006 indicates that system expansion is the preferred method for treatment of co-products within an LCA. The sensitivity analysis conducted in chapter 5 however had indicated the uncertainties that could be introduced both through the different options for replacement of livestock fodder by the co-products concerned and the inconsistency of boundaries for the background systems used for displacement.

Allocation by economic value was therefore retained as the preferred method of allocation, although since the economic value of the OB product and its co-product stream was uncertain, additional models were developed using mass allocation, such that a comparison of the novel and conventional technology was performed on the basis of the two different allocation methods, as well as no allocation at all.

In addition to the increased certainty of allocation factor values provided through using mass as a basis for allocation, the data obtained through this analysis could also be used to supplement the examination of the impact of using different allocation methods performed in chapter 5 and further discussed in section 9.2.

The results from this comparison, which identified the five most burdensome impact categories identified through endpoint normalisation as consistent with previous analyses were discussed at the end of chapter 8, but are amalgamated and reproduced in table 9.1-2, for ease of review.

Table 9.1-2: Top five characterised midpoint results for oil-body mayonnaise and conventionally produced mayonnaise using different allocation approaches

		Rapeseed system					
		No allocation		Economic allocation		Mass allocation	
		Oil-body mayonnaise	Regular mayonnaise	Oil-body mayonnaise	Regular mayonnaise	Oil-body mayonnaise	Regular mayonnaise
ALO	m^2a	13410	7106	7219	5210	3118	2992
FD	kg oil eq	1129	724	733	624	470	507
CC (CFP)	kg CO_2 eq	5693	3331	3364	2694	1821	1949
TET	kg1,4-DBeq	253	129	134	92.5	54.9	49.9
PMF	kg PM_{10} eq	13	7.6	7.4	6.0	3.8	4.2
		Sunflowerseed system					
		No allocation		Economic allocation		Mass allocation	
		Oil-body mayonnaise	Regular mayonnaise	Oil-body mayonnaise	Regular mayonnaise	Oil-body mayonnaise	Regular mayonnaise
ALO	m^2a	32525	16865	18202	12539	7260	6754
FD	kg oil eq	1069	730	722	630	457	496
CC (CFP)	kg CO_2 eq	6118	3646	3734	2955	1913	2033
PMF	kg PM_{10} eq	9.1	5.6	5.6	4.7	2.9	3.4
TET	kg1,4-DBeq	23.3	12	13.0	8.9	5.2	4.7

ALO = agricultural land occupation; FD = Fossil depletion; CC = Climate change; hh = Human health; es = Ecosystems; PMF = Particulate matter formation; TET = Terrestrial eco-toxicity.

Whilst the analysis of both the unallocated model and that using economic allocation indicated that the CFPs and wider environmental loads of the novel system were higher than those of the conventional process in all categories, analysis using mass allocation painted a different picture.

Comparison of the two systems modelled using mass allocation indicated that the novel process had a smaller CFP and improved credentials over the existing process in half of the impact categories, including three of the top five for the rapeseed system and four of the five for the sunflower system. However, the use of mass as an allocation parameter had the effect that a low product yield from seed was rewarded by lower impact results, since a reduced proportion of inventory was attributed to the product, rather than its co-product. Clearly the use of such data for decisions regarding the viability of the novel process or future optimisation for such a process would lead to different outcomes than when using the data developed using economic allocation. This will be discussed further in 9.2.1.

9.1.3. **Optimisation of the process**

Processes that are still at the laboratory stage such as the oil-body extraction process detailed and assessed in this thesis, have not generally been subject to any optimisation efforts and as noted by Tufvesson et al. (2013) comparison of such processes against well-established optimised conventional techniques can give misleading results.

The conventional processing routes against which the novel material was compared has, as outlined by Dijkstra (2009) been subject to process optimisation efforts, mainly targeted at mechanical efficiency and Karkani et al. (2013) note that traditional oil extraction is characterised by low-cost and high extraction efficiencies. In addition, most modern industrial processes generally involve some form of heat recovery, energy integration and recovery and recycling of materials within the process, all targeted to improve process efficiencies, reduce costs and conform to environmental permit requirements.

As an emerging technology, the novel process investigated here has undergone none of this optimisation at this point and the results presented in this thesis, which represent entirely original knowledge, must therefore be viewed with that in mind. Whilst the results of the analysis using mass allocation must be viewed with a clear understanding of its disadvantages and limitations for this system, the impact scores indicated were very positive for the viability of the process. They indicated that in its present state, its environmental credentials out-performed the conventional technology in most of the prominent impact categories, including carbon footprint, with very modest yield increases of a couple of percent required for parity in all areas.

Although the analysis of the OBM using economic allocation was less positive, indicating higher levels of burdens in all impact categories, the analysis presented and discussed in chapter 8 clearly showed that if optimisation efforts were targeted at improving the yield of wet oil-bodies from seed, the environmental performance of the process could be made equal to that of conventional technologies, even before optimisation through heat, energy and material integration occurred.

The required yields of between 27.5% and 32% (35% and 41% dry basis) were well within that theoretically possible for both seeds. Rapeseed has an oil content of around 40% (www.fediol.eu, 2013 (e)) which would translate to a maximum dry basis yield of 78.4%. Dry basis yields of this order of magnitude were reported during aqueous extraction trials for hemp oil-bodies, performed as part of the DTI funded 'SIPOS' project (Pers comm. Gray, 2014). The analysis presented here therefore demonstrates the real potential of the OB process for impact improvements, with yield performance at that level leading to the CFP of sunflower OBM reducing to 79% of the conventional mayonnaise value at the 50% dry basis yield level and 61% of the mayonnaise CFP at the 77% level.

It should be noted that the success of the OB process in achieving lower environmental impacts will not only be dependent on the ability to maximise the yield of OBs from the seed, but also dependent on the ability to utilise the residue as a co-product rather than a waste stream, since the use of allocation for the LCA is based on the understanding that the residues produced will be used as saleable by-product.

Whilst the work detailed within this thesis has not investigated the costs involved, it is apparent that the oil-body process needs some increased efficiencies to be environmentally competitive across the range of LCA analyses. Such developments may also reduce the costs through lower raw material and processing costs. Mindful of comment by Karkani et al. (2013) concerning the low-cost and high extraction efficiencies of conventional oil extraction processes, it may also be beneficial to include some form of life cycle costing in any future development work.

9.2. DETERMINATION OF THE IMPACT OF METHODOLOGICAL CHOICES

9.2.1. Treatment of co-products

Due to necessity to conduct multiple LCAs for the comparisons required for this research, attributional LCA was used for the analysis, with allocation of co-product burdens according to economic value. Given the considerable debate concerning the treatment of co-products within LCA however, and in-line with the requirements of ISO14044:2006 an investigation was conducted into the impact that different methods would have, including system expansion, economic and mass allocation, to provide information that could be of use both within this research and to inform the wider LCA community. Curran (2007), Halleux et al. (2008) and Morais et al. (2010) all indicate that allocation can have a profound effect on the results generated by an LCA and this proved to be the case with the analysis conducted here.

When assessing the seed oil systems, allocation by mass yielded values below those calculated using economic allocation, and system expansion provided results that were both higher and lower than the economic allocation values, dependant on the assumption concerning the livestock fodder to be displaced by the co-product streams. Use of alternative fodders were found to alter the results generated such that the normalised endpoint values varied from 62% higher than the results obtained via economic allocation to 32% below. Considerable uncertainty would be therefore introduced through using different fodder displacement options for the system expansion model, uncertainty that would be compounded by ambiguity and lack of transparency concerning the boundaries for the data-sets utilised.

It was therefore concluded that economic allocation, as used by the majority of oilseed LCAs accessed (Bernesson et al. 2004; Schonfield and Dumelin, 2005; Narayanswamy et al. 2005; Reijnders and Huijbregts, 2008; Stephenson et al. 2008; Nilsson et al. 2010; Stephenson et al. 2010; González-García et al. 2013) was preferable to system expansion.

For completeness, since the use of mass allocation did not contain the same uncertainties as system expansion, a change from economic to mass allocation was also assessed within the analysis of the OBM system. The impact that allocation method had when analysing the OBM systems and comparing with current technology firmly indicated the potential danger of using a single allocation method for LCAs upon which decisions or choices will be made.

Figure 9.2-1 uses the data from table 9.2-1 to indicate the percentage difference in characterised midpoint impact between the mayonnaise systems and their oil-body produced counterparts. From this, the large differences produced through allocation using a different basis is evident with the system using mass allocation showing the novel process to have improved credentials in most of the impact areas.

It is evident that for this comparative analysis, the use of mass allocation alone would potentially lead to different decisions being made than those based on results generated through economic allocation. It is therefore concluded that whilst attributional LCA may be the most suitable vehicle for analysing novel processes due to the required speed of analysis and probable necessity for multiple LCAs, sensitivity analyses concerning the allocation parameters and methods is imperative to ensure decisions are taken based on adequate information.

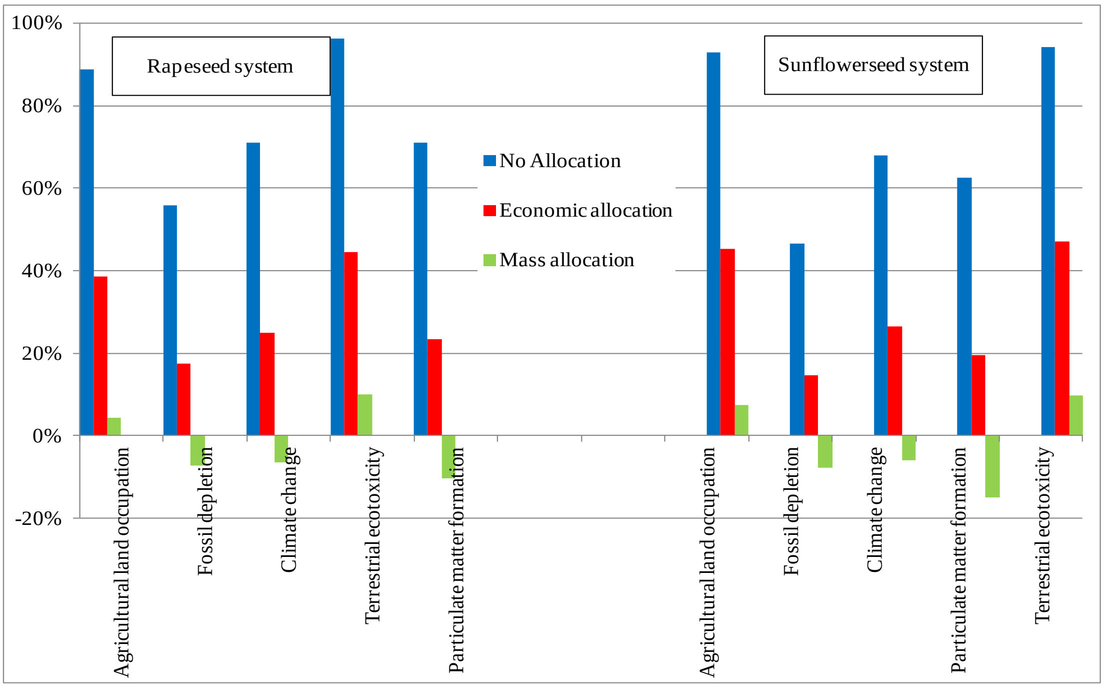

Figure 9.2-1: Percentage difference in characterised midpoints between mayonnaise and OBM systems with alternative allocation

9.2.2. **Normalisation**

Whilst normalisation, which is stated as optional within the ISO standards can provide a useful indication of the relative importance of different impact values within the system, it has a high degree of subjectivity and should not be taken as absolute. Transparency of reporting is an important requirement for any LCA and the level of differences shown through the comparative analysis of normalisation sets performed in chapter 5 showed that wherever normalisation is performed, details of the basis must be provided when viewing LCA results. Every effort should be made to ensure that LCA results are transparent such that readers could reproduce the work in the future or to facilitate a more direct comparison.

The process of normalising at the midpoint level attempts to address the transparency issue and the source of this data is readily available for the midpoint normalisation values (Wegener Sleeswijk et al., 2008). However as indicated by Van Hoof et al. (2013), the incompleteness introduced through the number of missing characterisation factors for different chemicals can lead to a greater level of uncertainty with midpoint normalisation. This can be particularly evident for toxicity categories where the number of substances characterised is so high (Laurent et al., 2012), with Wegener Sleeswijk et al. (2013) stating that the differences in global and European normalisation figures for the toxicity categories emphasise the relatively high uncertainty of normalisation factors for toxicity in both Europe and the world.

As discussed throughout the thesis, the approach advocated by Van Hoof et al. (2013) of identifying significance through endpoint normalisation but viewing the characterised results was used, since endpoint normalisation is less sensitive to uncertainties than midpoint normalisation through the parameters used being developed per damage category rather than per impact. For both endpoint and midpoint normalisation however, the normalisation factors available within ReCiPe(2008) for 'World' and 'European' values have large differences and therefore produce considerably different results. The comparison of seed oil results using European and Global normalisation datasets demonstrated that for the impacts at both the mid and endpoint level, prominent changes occurred in both the absolute values and relative ranking of the normalised results through moving from one normalisation set to the other.

These results confirm that LCA practitioners need to be aware of the size of differences that ensue through use of different normalisation sets and be particularly diligent at not only maintaining a consistent approach within comparative studies, but ensuring transparency within any reported results, to any audience.

9.2.3. **Use of CFP for targeting process improvements**

Given the sustained focus on greenhouse gas reductions that have led to an increased uptake of the single issue LCA variant CFP for reporting of environmental performance, the suitability of CFP as an environmental indicator was investigated with respect to mayonnaise, wet oil-bodies (WOB) and the novel emulsions.

In each of the systems studied, whilst climate change was not identified as the most prominent impact category, analysis indicated that those processes that contributed most to the CFP were also the highest contributors to the other most prominent impact categories. In their analysis of CFP as an acceptable metric based on correlation against other impact categories, Laurent et al. (2012) stated that '*a genuine correlation between carbon footprint and all other impact indicators can be observed if and only if all impacts from the product lifecycle predominantly stem from one or a few processes that covary'*. This would appear to be the case within the processes examined in this thesis, where the dominant impact contributions arise from cultivation.

From the analysis performed, it was therefore concluded that CFP and full LCA data yielded consistent results with regards to the most beneficial areas to target for further reducing environmental impacts. In this instance, the use of the single issue LCA variant would not lead to burden shifting within the system and CFP could be taken as an effective indicator of environmental performance.

LCA results are often difficult to interpret, as noted by Weidema et al. (2008) who state that LCA '*is often complicated stuff, and it is difficult to communicate and frequently hard to make clearcut decisions from'*. It is therefore advantageous to be able to focus on a limited number of parameters. Whilst the analysis performed in chapter 8 concentrated on the five most prominent impact categories, the confirmation that CFP is an effective indicator of environmental performance for this system will enable future researchers to use a single measure for targeting process improvements for this system. However care should be taken that any future optimisation involving prominent alterations to the process should be subjected to a full LCA to ensure consistency and provide further confidence that burden shifting is being avoided

9.3. MODELLING THE NOVEL PROCESS AT A COMMERCIAL SCALE

It is estimated that about 80% of all environmental effects associated with a product are determined in the design phase of development (Tischner, 2000), and use of LCA at this stage for any product or process can be key to unlocking the environmental improvement potential, forming the basis of eco-design (Hetherington at al. 2014). There are a number of methodological and practical difficulties that arise from using LCA at this stage however which Hetherington at al. (2014) broadly categorise as 'system boundaries', 'data availability', 'scaling issues' and 'uncertainty'. Each of these was encountered during the research contained within this thesis.

9.3.1. System boundaries

Correct specification of system boundaries is key to achieving the functional equivalence required for effective comparisons between LCA studies, as stressed within Hospido et al. (2010) who suggest that for comparative studies; only the part of the production chain that is affected by the change in production technique is included within the system boundary. However, as with the systems analysed here, such simplification is not always possible.

The oil-body emulsions produced could not be compared against their equivalent oils, since they were not designed as direct replacements for oil, but to replace a product produced from the oil. This is often the case, as new processes and materials will not necessarily be direct replacements for their existing counterparts and their inclusion within an established process may frequently entail process or procedural changes within the process or product system to be used. In cases such as this, the material needs to be compared as part of a wider product system, in this case the manufacture of mayonnaise, to ensure functional equivalence.

9.3.2. **Scaling issues**

Gathering process data for any LCA can be problematic, however acquisition of such data for novel processes that are still at the laboratory stage poses particular problems. Several researchers discuss the use of LCA for innovation and product development including Arena et al.(2013), Munoz (2006), Piekarski et al. (2013) and Hospido et al. (2010), who outline specific elements that are pertinent to LCA development for novel food products and discuss the potential for use of simulation techniques to acquire such data.

The impact of using alternative data collection strategies for modelling the lab-scale process was demonstrated within chapter 7, where the comparison of LCA results generated using the three alternative approaches generated very different results. This supports the comments of Tufvesson et al. (2013), who following a wide ranging review of articles concerning LCAs for chemical products as part of their assessment of LCA in green chemistry, commented that where LCAs performed on emerging technologies were compared with LCAs for well established processes, this could lead to misleading results.

Through analysis of these findings it was concluded that whilst LCAs could be developed using mass balance and energy data collected from laboratory test runs, these would not be comparable with industrial scale processing and whilst they would be suitable for limited hot-spot analysis, they were inappropriate for comparative LCAs with commercial scale processes.

A commercial scale proxy was developed through combining laboratory acquired mass balances with manufacturer's data for energy consumption. This was deemed suitable since the mass balance data reflected the state of the current process and the energy data represented usages in an industrial setting. Since the properties and geometry of the seed would remain the same as the process was scaled up, it was assumed that the ratio of bicarbonate solution to seed would also remain the same, which entailed a linear scale-up of mass-balance data. This approach was used successfully throughout the research and the step-wise procedure advocated in chapter 7 will provide useful methodology for any future work in this area, in addition to being transferable for use by other researchers and designers within their LCAs for novel process development.

The acquisition of raw primary data from any future scale-up operations would however be of enormous benefit, both to confirm the environmental performance of the system, and improve understanding of the relationship between projected and actual commercial scale data. If the oil-body process progressed to pilot and semi-commercial scale, the importance of collection of comprehensive mass and energy data should be stressed to the research team involved.

9.3.3. **Data availability**

Beyond the issues encountered accessing representative mass and energy data for the OB process, availability of LCI data for all material inputs is a problem that is exacerbated for early stage LCA work such as this. As noted by Hetherington et al. (2014) developing process studies often involve multiple LCAs and a speed of assessment that requires information to be provided at the appropriate stage for process changes to be most effectively introduced. Data gathering is one of the issues raised by Hospido et al. (2010) who recommend that specific data should be utilised for the foreground system, whilst average data – with a suitability check, used for the background system.

Use of secondary data is often the only practical solution, since primary data would either not be available or take too long to gather. As detailed in the relevant sections, the majority of secondary data used within this modelling was accessed via the Ecoinvent and LCAFood databases that are available via SimaPro. However, as noted by Jimenez-Gonzalez et al. (2000) LCA databases represent just a part of the raw materials used in chemical and biochemical companies and as with many LCAs there were instances in this research where data for previously undocumented materials was required and time was not available for collection of primary data, even if a source could be identified and agreed.

No data could be accessed for Peroxy Peracetic acid (PPA) which was used in the laboratory trials for controlling microbial carry-over from the seed into the OB material (Khosla, 2011), however if the OB process were to be commercialised, this process stage would almost certainly be replaced by a pasteurisation unit. Rather than investing time in accessing data for a step that would not be required therefore, LCI data was instead sought and found for pasteurisation.

Despite being a well known household commodity, no data could be found for sodium bicarbonate either, and as this material was an integral part of the process which would continue to be used at the commercial scale, data which was as similar to the bicarbonate as possible was sought and an aggregate proxy dataset developed.

Judgements and assumptions such as these must be taken to overcome the challenges of data collection that an LCA study on a developing process poses.

9.3.4. **Uncertainty**

All LCA studies have a degree of uncertainty that stem from data and assumptions used, together with inherent uncertainties with certain elements of the methodology such as Life Cycle Impact Assessment or allocation.

These are often compounded for LCAs on emerging processes by aspects such as unknown future applications, unknown industrial scales, and the degree of technology development, each of which had an effect on the uncertainty of the LCAs developed within this thesis. Assumptions concerning both future applications and scale of operation have had to be made to develop the required models and whilst all such assumptions were justified and documented for transparency, they undoubtedly have the potential to introduce a relatively large level of uncertainty. The sensitivity of the results to these factors can only truly be quantified through the development of entirely new LCAs for comparison, for example of the use of OB within an alternative product, or a larger capacity production plant which necessitates alternative equipment.

Further sources of uncertainty distinct to this process, but potentially shared by all LCAs on novel processes include the use of scale-up strategy and choice of technology adopted. The findings of analysis of the effect of different scale-up strategies were documented in chapter 7 as part of the process whereby the decision to adopt the commercial scale proxy was taken. Whilst indicated as the best option for modelling, the use of the commercial proxy will also potentially introduce large inaccuracies, the true level of which can only really be quantified by the collection of representative data for comparison at the larger scale. Likewise, the assumption of linear scaling for the mass balance data may be questioned if different mixing profiles required the larger volumes would affect the action of the soaking medium.

Those areas deemed to have the potential to introduce the largest uncertainties were investigated by using sensitivity analyses and reported in the following chapters:

- Alternative cultivation datasets – chapter 5
- Modified transportation data – chapter 5
- Different normalisation datasets – chapter 5
- Different scale-up data acquisition strategies – chapter 7
- Alternative methods for treatment of co-products - chapters 5 and 8.

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CHAPTER 10. CONCLUSIONS

The central question to be answered by this research was whether the novel processing route for production of food grade edible oil emulsions via aqueous extraction of oil-bodies had a better environmental profile than that of the existing technology route. In other words, was the oil-body material more "environmentally friendly" than its conventional counterpart?

In order to meet this aim, life cycle assessment (LCA) was used to develop a functionally equivalent comparison of an emulsion product formulated with oil-body material and a standard food emulsion product. To facilitate this, the research was focussed on six key objectives, designed to both identify the environmental loads of the systems and scrutinise the impact of number of methodological choices for LCA. These objectives involved the creation of a set of LCAs for four separate categories of product system, encompassing seed oils, mayonnaises, aqueously extracted oil-body materials and mayonnaise-like oil-body emulsions.

Analysis of these models in line with the requirements of the objectives, enabled conclusions to be drawn that resulted in original knowledge development in several areas. Details of the key findings will be presented within this chapter, together with a summary of the original contributions that they provide. In addition, a brief commentary on the issues and limitations of the research will be provided, together with some recommendations for further work, based on the findings of the research.

10.1. KEY FINDINGS

The conclusions drawn from the analysis and synthesis of the results generated can be summarised in six key findings, these are detailed below:

1. The novel processing route for production of food grade edible oil emulsions via aqueous extraction of oil-bodies has clear potential for improved environmental performance over current technology, although in its pre-optimised state, this is dependent on choice of allocation method.
   1.1. When assessed using economic allocation, rapeseed mayonnaise-like oil-body emulsion has impact values of between 17% and 45% higher than those of conventionally produced mayonnaise, including a 25% higher carbon footprint (CFP). The comparative values for sunflower were 17% and 39% with the same magnitude of increase for CFP
   1.2. When assessed using mass allocation, impact values of the rapeseed mayonnaise-like oil-body emulsion were +/-10% of the values obtained for conventionally produced mayonnaise, including a 7% lower CFP. The comparative values for

sunflower were 15% lower to 10% higher than its conventional counterpart, with a 6% reduction on CFP

1.3. The use of sensitivity analysis for different allocation types is absolutely imperative for novel processes and wherever development decisions are to be taken on the basis of LCA results.

2. Impacts from seed cultivation provide the highest contribution to environmental load in each of the systems analysed (seed oils, mayonnaises, wet oil-bodies and mayonnaise-like oil-body emulsions). Thus optimisation of the novel processing with respect to yield of oil-bodies from seed is key to maximising the environmental gains for the novel process.

   2.1. When assessed using economic allocation; for parity with conventional mayonnaise processing, the yield of wet oil-bodies from seed needs to increase from the current 21.7% (27.7% dry basis) to between 29% and 32% (37% and 41% dry basis) for the rapeseed system and 27.5% to 31.5% (35% to 40% dry basis) for the sunflowerseed system dependent on environmental impact. For the CFP of the novel system to equal that of the conventional processing, the wet oil-body yield from seed needs to increase to 29.5% for both seed types, equivalent to 37.6% dry basis yield

   2.2. Very modest increases in oil-body yield from both seed types from the current 21.7% (27.7% dry basis) to 23% to 24% (29% dry basis) are required for the mayonnaise-like oil-body emulsion to equal or better the environmental loads in all key impact categories when assessed using mass allocation.

3. CFP results provide consistent data to the LCA results when indicating the processes with the highest contributions to each of the most prominent impact categories for each of the product systems analysed.

   3.1. Carbon footprinting can be used as an effective indicator of environmental performance for each of the systems assessed

   3.2. Targeted optimisation of the oil-body extraction process, based on data from CFP analysis would lead to improvements throughout all of the most prominent impact categories.

4. Each of the process modifications investigated provided environmental improvements to the system, with processing seed at a higher moisture content yielding improvements of between 0.7% and 5.4% dependant on seed type and impact category, processing the material at the farm providing impact reductions of 3.7% to10.4% and removing the sodium bicarbonate pre-soaking stage yielding improvements of 3.0% to 4.9%.
   4.1. The modest nature of those reductions coupled with the larger potential arising from extraction yield optimisation make only the modifications that are simple to implement feasible to pursue at this early development stage.
   4.2. As a simple modification to reduce impacts further, providing its removal does not lead to increased requirements elsewhere, the pre-soaking of seed in water, rather than sodium bicarbonate solution should be further investigated.

5. When analysing a process in the early developmental stage, utilisation of laboratory sourced data for construction of an LCA would be suitable for limited hot-spot analysis, but is inappropriate for use in comparative LCAs with commercial scale processes.
   5.1. Use of a commercial projection of the novel process, following the step-wise procedure advocated, that combines laboratory acquired mass balances with manufacturers data for energy consumption has been demonstrated to be an effective method for acquiring LCI data, providing uncertainties are acknowledged. This could be equally well utilised for other emerging processes.

6. Large differences occur in the magnitude and relative significance of normalised impact results when changing from European to World normalisation at both the mid and endpoint level.
   6.1. LCA practitioners need to be aware of the size of differences that ensue through use of different normalisation sets and be particularly diligent at not only maintaining a consistent approach within comparative studies, but ensuring transparency within all reported results, to any audience.

10.2. **KNOWLEDGE CONTRIBUTION**

For the future development of the aqueous oil-body extraction process, the findings of the research presented in this thesis will provide important information to researchers and developers. This not only confirms the environmental potential of the process when compared against conventional technologies, but will enable the correct targeting of optimisation efforts. The conclusions reached can be used by both by researchers within the SEIBI consortium to assist with the future development of the oil-body extraction process and any other individuals progressing the aqueous extraction of oil-bodies with a view to scaling up to commercial production.

In addition to generating the environmental profiles required to fulfil the research aim and objectives, the construction and analysis of the LCA models required, enabled the generation of original knowledge through the quantification of impacts for a range of processes that had either not previously been assessed or for which no published data could be found. Good quality, transparent data is essential for creation of any LCA and the publication of such information through this thesis will therefore be of use to others in the wider research field involved in LCA construction.

Similarly, whilst the use of rape and sunflowerseed oils within products has been the subject of a considerable amount of previously published LCA work, particularly in the bio-fuels area, the few studies that have reported the results of LCAs on the seed oils as products have used different methodological choices within their modelling. As such, the reporting of the results for the attributional, cradle to gate LCA performed here, using economic allocation, will provide valuable additional data on the environmental impacts of rape and sunflowerseed oils, for use by and within the edible oils and LCA community.

The results, discussion and conclusions from each of the analyses concerning methodological choices will be of interest to LCA practitioners and developers in assisting the improved understanding of the sensitivities, limitations, problems and potential with LCA, through examining the impacts of normalisation and allocation in a developing process context. Furthermore, the step-wise procedure for applying LCA to lab-scale processes, to construct a model of the process at the commercial-scale will be of considerable use to process and product developers in utilising LCA at the earliest developmental stage. This will maximise the potential for eco-design.

In addition, the examination of the extent to which the CFP could be used as an effective environmental indicator for these processes will not only be constructive for the SEIBI project team as an indication that the simplified single-issue approach can be used for ongoing development work, but will also provide useful, analysis based information on the applicability of CFP for environmental assessment to the LCA community.

10.3. ISSUES ENCOUNTERED AND LIMITATIONS OF THE RESEARCH

With any extended research project such as this, it is important to reflect on any limitations that exist within the analysis, results and conclusions presented, to ensure that future readers are able to view the research findings in a transparent manner.

The acquisition of suitable data for the construction of the LCI is challenging for any LCA study. Primary data, specific to the individual process being assessed is always preferable for the foreground system, however this cannot always be acquired. For the processes involved within this research, all data for the conventional processes was sourced from literature or proprietary databases apart from the aggregate data supplied by the mayonnaise manufacturer for energy and water consumption at their facility. To address this issue, sensitivity analyses were conducted throughout the investigation to investigate the impacts of changing key datasets and assumptions.

Whereas the initial projected timelines within the SEIBI project proposed that scale-up trials for the oil-body extraction would take place in August 2011, the speciality processor that would have performed these trials withdrew from the process and no replacement was found. As such, it was not possible to acquire primary data for the aqueous extraction on a semi-commercial, pilot scale and therefore as detailed in chapter 7, modelling assumptions and projections of data were necessary. Whilst this was disappointing, it did prompt the examination of the issues with regards to obtaining and generating data at a suitable scale for novel LCAs, for which supplementary information, when available would be beneficial, as will be discussed in the recommendations for further work.

10.4. RECOMMENDATIONS FOR FURTHER WORK

From the results and conclusions generated from this research, a set of recommendations can be made, concerning both the future development and further assessment of the oil-body extraction process.

Having determined that the process for aqueous extraction of oil-bodies has definite potential for improved environmental performance over its conventional counterparts in the areas examined, further yield optimisation work should be performed to maximise its early potential. If the oil-body extraction process was to proceed to pilot scale and beyond, the collection and documenting of mass and energy balance data should be performed. This would not only provide essential information for further analysis of environmental performance, but augment the understanding of the relationship between data collected at lab scale, the larger pilot scales and eventual commercial scale. This would be invaluable to assist with the development of LCA for new technology assessment.

The work detailed within this thesis has indicted that the novel process can rival the environmental burdens of the existing process route and potentially offer substantial environmental improvements, if optimisation were to be performed. The economies of scale exploited by the traditional oil extractors mean that conventional oil-processing is characterised by low-cost and high extraction efficiencies. Whilst the LCA generated using commodity prices for allocation indicated that the oil-body extraction process needed to increase efficiencies to be environmentally competitive in many impact areas, for the process to progress to scale-up and beyond, some form of life cycle costing would be recommended.

Whilst the examination of the effectiveness of carbon foot-printing within this study proved that CFP could be used as an effective environmental indicator for this system, any future optimisation involving prominent alterations to the process should be subjected to a full LCA to ensure consistency and provide further confidence that burden shifting is being avoided.

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APPENDIX A. PUBLISHED PAPERS

The following papers are produced within this appendix:

Hetherington A C, Li Borrion A, Griffiths O G, McManus M C. (2013). Use of LCA as a development tool within early research: challenges and issues across different sectors. *International Journal of Life Cycle Assessment.* On-line.

Author accepted proof:

Use of LCA as a development tool within early research: challenges and issues across different sectors

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Abstract

Purpose The aim of this paper is to highlight the challenges that face the use of life cycle assessment (LCA) for the development of emerging technologies. LCA has great potential for driving the development of products and processes with improved environmental credentials when used at the early research stage, not only to compare novel processing with existing commercial alternatives, but to help identify environmental hotspots. Its use in this way does however provide methodological and practical difficulties, often exacerbated by the speed of analysis required to enable development decisions to be made. Awareness and understanding of the difficulties in such cases is vital for all involved with the development cycle.

Method This paper employs three case studies across the diverse sectors of nanotechnology, lignocellulosic ethanol (biofuel), and novel food processes, demonstrating both the synergy of issues across different sectors and highlighting the challenges when applying LCA for early research. Whilst several researchers have previously highlighted some of the issues with use of LCA techniques at early-stage, most have focused on a specific product, process development, or sector. The use of the three case studies here is specifically designed to highlight conclusively that such issues are prevalent to use of LCA in early research irrespective of the technology being assessed.

Results The four focus areas for the paper are; system boundaries, scaling issues, data availability and uncertainty. Whilst some of the issues identified will be familiar to all LCA practitioners as problems shared with standard LCAs, their importance and difficulty is compounded by factors distinct to novel processes as emerging technology is often associated with unknown future applications, unknown industrial scales, and wider data gaps that contribute to the level of LCA uncertainty. These issues, in addition with others that are distinct to novel applications, such as the challenges of comparing laboratory scale data with well established commercial processing, are exacerbated by the requirement for rapid analysis to enable development decisions to be made.

Conclusions Based on the challenges and issues highlighted via illustration through the three case studies, it is clear that whilst transparency of information is paramount for standard LCAs, the sensitivities, complexities and uncertainties surrounding LCAs for early research are critical. Full reporting and understanding of these must be established prior to utilising such data as part of the development cycle.

Keywords: Life Cycle Assessment, biofuel, nanotechnology, food processing, novel, emerging technologies, scale-up

1. Introduction

As a tool designed to quantify the full range of environmental impacts within a system, LCA has traditionally been undertaken retrospectively, using data from existing large scale processes. However great potential for environmental improvement exists using LCA within the design stage of any product or process where it is estimated that about 80% of all environmental effects associated with a product are determined in the design phase of development (Tischner, 2000) . Indeed determining where improvements can be made whilst a process is still at the laboratory stage can be key to unlocking the environmental improvement potential, forming the basis of eco-design. Its use through the more generic life cycle thinking is also encouraged through numerous policies and legislation, such as those based on producer responsibility (eg EU Directives such as the WEEE Directive (EC, 2006), End of Life Vehicle Directive (EC, 2003)) and those that promote the use of aspects of LCA such as the Renewable Energy Directive (EC, 2009).

Whilst a myriad of methodological challenges are debated within the LCA community (Ekvall and Weidema 2004; Roy et al. 2009), there is a general consensus on LCA's suitability as an effective tool for determining environmental performance (Finnveden et al. 2009) and it is used widely as a decision-making tool in process selection, design, and optimization (Del Borghi et al. 2007). Koller et al (2000) and Tufvesson et al (2013) note that full-scale LCA is often thought of as too difficult or time consuming to pursue at the research or development stage of a new product or process. There are certainly a number of methodological and practical difficulties that arise from using LCA at this stage and Kunnari et al (2009) discuss options for methodological changes, based on the work of Nielsen and Wenzel (2002) who advocate the use of a stepwise LCA procedure in parallel with the development process. Use of LCA in this way often entails the assessment of lab and/or pilot-scale processes to generate environmental load data, which can then be used to optimise the developing process. This data may also be used to compare with existing industrial processes, to demonstrate or identify the environmental advantages of the 'novel' process over the existing activities.

Within all LCA's, the clear stipulation of goal and scope is essential, however for emerging LCAs several elements require particularly careful attention. Clarity on the intended use of the output and the anticipated target audience need especially careful definition to ensure that methodological choices are correctly made and results reported in a manner appropriate to the needs. As will be demonstrated within the case studies discussed here, the differentiation of purpose has prominent ramifications for methodological choices which are exacerbated for early-stage LCA and information on whether the study is for 'hot-spot' identification or comparison with existing processes, together with whether the results are purely for internal use or future external publication must be agreed by all stakeholders at the outset..

For an appropriate and detailed LCA in practical decision-making, a wealth of information is required, which might be hard to obtain within the early phase of process design. Whilst inventory data collection for existing processes may be arduous, the task is exacerbated for lab-scale processes, with issues such as the use of unfamiliar and/or novel materials, prominent differences in laboratory methods and equipment compared with those on an industrial scale and processing issues that differ from those at a larger scale. Wider topics that can be investigated within an 'early-stage' LCA are the exploration of many alternative pathways for the future, with features including diversity in feedstocks, fuel composition, and by-products. Emerging technologies and novel products are often prominently different from the established materials or processes they aim to replace, with operational, in-use and disposal data all likely to differ. LCAs at this stage therefore pose a multitude of challenges due to scale issues and technology uncertainties, which make choice of functionality for assessment problematic.

The purpose of this paper is to highlight the methodological issues and complexities concerning the integration of LCA for early research, spanning differing technological spheres, through the collation of experience from case studies in three completely different sectors: nanotechnology, lignocellulosic bio-fuel, and novel food processing. Whilst researching the environmental impacts within these different areas, the authors identified many commonalities in the challenges and issues encountered, some of which, whilst similar to those encountered in standard LCA's, became more prominent and critical due to the requirement for speed of assessment for 'novel' technologies. Kunnari et al (2009) note that '*simplification of LCA cannot be avoided in the development of new products*', however even when simplified, using LCA for assessment of emerging technologies brings in complexities that must be acknowledged and understood by all stakeholders to enable effective development decisions to be made. The main issues discussed in this paper are comparability, scaling, data and uncertainties. Each of the emerging technologies discussed within this paper are within the laboratory stage, or very early stages of industrial pilot-schemes, and therefore LCA at this stage is key in order to ensure reduced environmental impacts, whilst expedience in providing results that are as representative as possible is paramount to support the required pace of development.

2. Case Studies

Each of the three case studies represent areas where there is increasing research interest and so offer good examples for the use of LCA at an early phase. Although diverse in nature, the experiences gained through using LCA to assess environmental impacts as part of the development process within each case study area illustrate that such issues are not technology dependant, but span different sectors and are common to early stage LCA studies. This supports commentary by (Nielsen and Wenzel (2002), Kunnari (2009), Tufvesson et al (2013), who reported similar challenges within their particular research areas. For each case-study, an overview is presented to enable work to be put into context.

2.1 Nanotechnology

Nanotechnology (the synthesis and manipulation of objects at the nanoscale, <100nm) is an emerging multi-disciplinary field. The inventory of consumer goods incorporating nanomaterials has increased by 521% since it the start of measurement in March 2006 (Woodrow, 2011); industrial applications are also being rolled-out at a similar rate of progress. Nano materials are found in numerous every day products, such as sun cream, antibacterial coatings, dirt-repellent and anti-crease textiles, and are used in medical imaging techniques. Despite increased understanding of the science and engineering behind nano-synthesis and likely nano-applications, very few published studies investigate the life cycle implications of nanomaterials (Bauer et al, 2008, Buchgeister et al. 2008; Gavankar, Suh et al. 2012; Kim and Fthenakis 2012)).

Carbon nanotubes are, arguably, the most established examples of engineered nanomaterials with one of the earliest reported synthesis routes (Ijima 1991), and a material with wide-ranging emerging and near-term projected applications. However, the production of carbon nanotubes has only recently moved from laboratory to industrial, pilot-scale levels, and the selection of the 'finalised' industrial process design is still under development (Zhang et al. 2011). Upadhyayula et al. (2012) recently reviewed the progress made in understanding the life cycle impacts of carbon nanotubes, concluding only 7 examples of LCA publications presently available, all of which relate to laboratory and small-scale synthesis of nanotubes. Similarly, a more recent literature search by the authors yielded in the region of 20 examples of a life cycle approach being applied to the assessment of nano-, manufacturing, materials, technologies other than carbon nanotubes (eg, Lloyd and Lave, 2003, Ju-Nam and Lead, 2008, and Kushnir and Sanden, 2011). The lack of life cycle information on nanotechnology is a matter for concern when attempting to quantify the holistic environmental benefits these materials may, or may not, deliver (Bauer et al. 2008; Som et al. 2010).

The impacts of nano-specific environmental effects are wanting from all published LCAs of nanomaterials. Despite scientific evidence purporting to potential, albeit largely unquantified, human health risks (Oberdoester 2010) and wider ecological impacts (Wiesner, Lowry *et al.* 2006), exact understanding and accounting of cause-effect and transport mechanisms of nanomaterials are still under-development (Rickerby and Morrison 2007; Peralta-Videa, Zhao *et al.* 2011). The lack of impact assessment methodologies to account for any potential 'nano-impacts' result in LCA studies only going so far as to measure the energy usage and bulk material and chemical consumption when assessing nanotechnology impacts (Bauer, Buchgeister *et al.* 2008; Gavankar, Suh *et al.* 2012; Kim and Fthenakis 2012).

2.2 Lignocellulosic biofuel

The use of bioenergy is promoted within the EU and UK through, for example, the Renewable Energy Directive (EC, 2009) and the RTFO (DfT, 2012). However, there has been much discussion surrounding the sustainability of bioenergy, especially focusing around the food versus fuel debate (Royal Society, 2008). For this reason second generation biofuels such as lignocellolosic biofuel are considered to be more beneficial than fuels made from crops that can also be used for food. With the focus on the sustainability issues surrounding biofuel, an increasing amount of published material in the area of biofuel LCA can be found, as outlined within Bessou et al (2011). Although LCA work (Kim and Dale 2006) has shown environmental benefits associated with lignocellulosic ethanol, most studies have focused on assessing the farming systems with a generic assumption of the ethanol conversion process; very few have addressed any specific environmental issues for the conversion process. This is due to process uncertainties and the non-availability of commercial plant (Spatati et al. 2010). Despite extensive research on lab and small scale within the scientific community, there is presently no large scale commercial lignocelluloses-to-ethanol facility. Thus, technology uncertainty and potential commercial scale operation parameters also contribute to the gap (Spatati et al. 2010).

2.3 Novel foods and food processes

LCA is an established tool for the assessment of whole-life impacts of food products, and Anderson and Ohlsson (1998) and Roy et al. (2009) provide information on the multitude and variety of LCA studies performed in this sector. In recent years however its popularity has soared with the increased focus on greenhouse gas (GHG) accounting over the entire supply chain fostered by such initiatives as the UK 'Carbon Label' and Sweden's 'Klimatmärkning'. Edwards-Jones et al (2009) note that '*in the future consumer and legislative responses to carbon labels may favour goods with lower emissions*' a statement which highlights the importance of using LCA techniques to optimise environmental performance of food production at the earliest possible stage of development.

Despite the popularity of LCA within food manufacturing, and the obvious requirement for studies at the earliest possible developmental stage, there is very little published literature concerning LCA of 'new processes' within food products or the challenges of performing LCA at this early stage. Pardo and Zufia (2012) reported on their study concerning LCA of food-preservation technologies and Hospido et al. (2009) discuss some of the methodological issues associated with performing LCAs over novel food products. The latter provides useful confirmation of some of the challenges identified with using LCA at this stage, with issues such as the inventory development stage, definition of functional unit (FU) and the assumptions required to estimate future developments and uses all being highlighted. They propose a recommended approach within five identified areas, namely 'type of LCA, functional unit, system boundaries, data gathering and scenario development' and advocate a check of its applicability to other industrial sectors.

3. LCA for early research

The majority of LCAs are traditionally performed at the pilot scale, where primary data can be readily acquired, or industrial scale when the process is mature and thus generates necessary detailed inventory data. As indicated in figure 1 however, for LCAs on emerging technologies there is no 'mature' plant available for data collection and a considerable amount of secondary and proxy data must be utilised.

Whilst the requirement for and variability of this data may reduce as the development progresses, the future potential for development of such plant may in part be dependent on the verification of improved environmental credentials at the earliest stage. Such LCAs are typically commissioned to provide information for a variety of stakeholders including project researchers, developers, and decision makers, which may be internal project managers, external project financiers or both. Practitioners of early stage LCAs must be sensitive to the increased levels of uncertainty that can be prevalent and ensure clarity on the intended and allowable use of results within the goal and scope. All information communicated must be commensurate with the needs of each stakeholder and the sensitivities or caveats of the study adequately explained to enable the recipients to appreciate the true nature of the results.

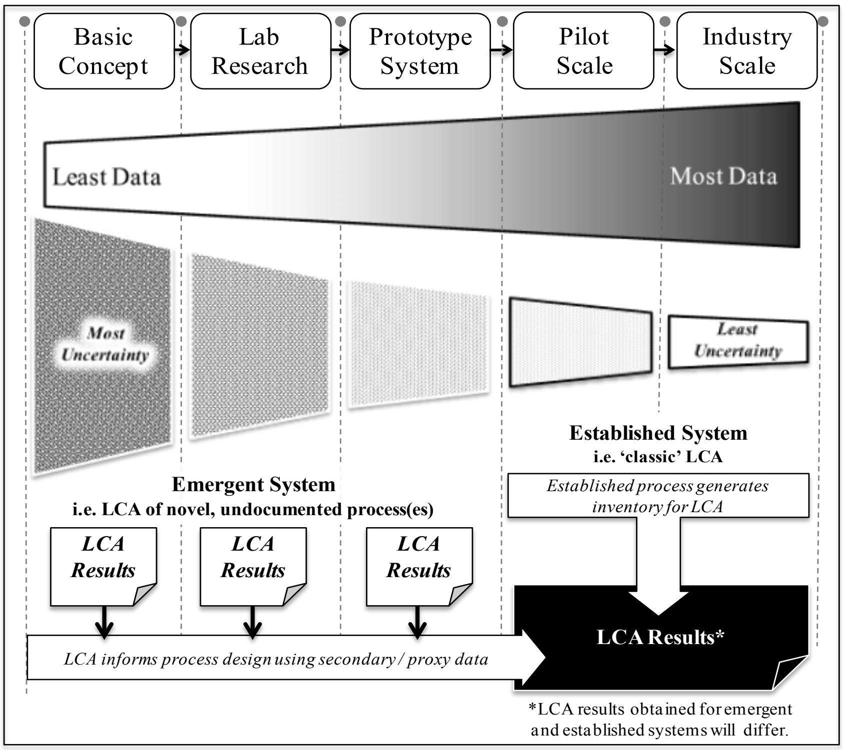

Figure 1 LCA at early research stage

3.1 Comparability

As previously highlighted, one of the objectives of performing LCA on emerging technologies can be to benchmark environmental performance against existing commercial products or processes. The problems of incomparable functional unit and system boundaries exist in all LCAs and are certainly not restricted to studies into novel processes, however, it is proposed that these problems are exacerbated when applying LCAs to early research and compounded by the required speed of assessment which is critical to enable development decisions to be taken in a timely fashion. Suh et al. (2004) confirms this, noting that choice of system boundary may have an influence on rankings in comparative studies, thus leading to incorrect conclusions and decisions about which products to promote. The function of the product may not be comprehensively defined, with systems prone to change when scaled up, processing stages may not be fully identified, co-product usage unclear and end of life treatments unknown, all of which can result in the exclusion of processes and life cycle stages from the system boundary. Such actions can lead to inadequate interpretation of the results and incorrect decisions being taken.

Rapid advancement in nanomanufacturing practices, likened to that seen by the semiconductor industry (Klöpffer 2007), see advancements in tooling and production techniques resulting in process cycle times of 18 months (Krishnan et al. 2008). When practices, and therefore associated manufacturing data, are subject to changes within such short time periods the comparability of studies becomes much more difficult. The functional unit for many cradle-to-gate traditional bulk materials within nanotechnology is often based on the mass of a formed product. However, when dealing with nanomaterials, dominant functional changes can occur from subtle alterations in the surface area, structure, and purity of the product (Daniel and Astruc, 2004). Thus nanomaterials require a greater level of technical definition to be stated for the actual product formed and its applicability to specific applications (Wender and Seager, 2011).

Functional equivalence is paramount, as stressed within Hospido et al. (2010) who suggest that for comparative studies; only the part of the production chain that is affected by the change in production technique is included within the system boundary. This suggestion would be compatible with the observations of Kunnari et al (2009) however such simplification is not always possible if functional equivalence is to be achieved. ‘New’ materials produced will not necessarily be direct replacements for their existing counterparts and as such will not be functionally equivalent as a stand-alone commodity. Their inclusion within an established process may often entail process or procedural changes within the process or product system to be used and the functional unit chosen must be able to reflect and encompass this. For example during early stage LCA of oil body extraction from oilseeds, the ’new’ ingredient could not be compared with the ingredient it had replaced, since the ‘new’ material possessed qualities and attributes that entailed the removal of several process steps and augmentation with others when incorporated into the production of an existing foodstuff. In this instance the material needed to be compared as part of a food product system to ensure functional equivalence. Simplification of boundaries was not possible if functional equivalence was to be assured.

Similarly, the system boundaries of lignocellulosic biofuel can vary from study to study depending on the inclusion or exclusion of some processes. For the same supposed system boundary, e.g. well to gate, in terms of ethanol conversion process, the actual boundaries are not always clear, and in some studies the processes used have not been specified (Borrion et al 2012). For example, among LCA studies published in this area, not all studies have taken account of chemicals, enzymes, nutrients, and the infrastructure such as equipment (MacLean and Spatari 2010). The decision to exclude certain elements of the process in the system boundary leads to problems, such as incomparability with similar studies and fossil reference systems. Functional equivalence may also be impossible to define when consumption patterns are altered by a new product, Bauer et al. (2008) suggest that in such cases the expected changes to the market and resultant effects on existing products need to be modelled. This links a more traditional attributional type LCA with a consequential LCA.

When applying LCA to early research, whilst speed of execution is important, information supplied to decision makers must contain clear statements and explanations of the complexities of the modelling undertaken. Clarity of purpose must be ensured within the goal and scope, with care taken to ensure that identical system boundaries are applied and functional equivalence is assured with any system used for comparison. Assumptions concerning future scenarios and technology development should be clearly labelled, functional units carefully selected and where appropriate, multiple functional units should be shown within studies to aid future comparisons. Whilst many of these aspects may not appear unique to early research LCAs, the way that the data may be used heightens their importance and makes clarity amongst all concerned essential.

3.2 Scaling issues

In order to conduct an LCA study one must gather inventory data. For ‘standard’ LCAs, this is typically industrial data from established processes, however this is clearly a problematic proposition for novel processes. Obviously, lab-scale processes do not entail the same level of complexity of equipment and commercial or industrial scale processes will almost certainly require additional processing elements such as material and heat transfer equipment (at the minimum) and entail the use of alternative processing equipment more suited to larger scale production. Conversely, at the lab-scale, processes may exhibit a far lower yield than would be possible in a commercial facility. For example, within a novel food processing project, the authors observed a lab-scale yield of approximately 10% when producing a particular material, however when this was transferred to pilot-scale for further testing, the larger scale equipment was able to attain yields in the region of 80%. Clearly, such a large discrepancy in the basic mass balance data would have an enormous impact on the overall results of an LCA, and the assessment of viability of the process.

In the absence of peer-reviewed life cycle inventory datasets for nanomaterials, achieving confidence in the suitability of data collected from a particular laboratory scale synthesis route can be difficult. There are often a multitude of alternative reported synthesis routes for any given nanomaterial. In such cases the LCA practitioner needs to establish, based on given technical, economical or other related information, whether a particular nanomanufacturing process is likely to continue onto further stages of industrial development. In the case of carbon nanotubes (CNTs), many synthesis routes exist, each have merits and give rise to different structures and properties, but with only a few pilot schemes producing CNTs worldwide (Zhang et al. 2011), the process most likely to be adopted for widespread industrial growth of CNTs is a matter of considerable uncertainty. Failure to keep abreast of current material production methods could potentially result in, at best, wasted effort and, at worst, a misrepresentative and wholly inaccurate LCA; counterproductive in achieving the objectives of forecasting the impacts of emerging technologies.

Manufacturing at increasingly smaller scales is proving to be ever more energy intensive (Gutowski et al 2007). Whilst efficiency gains are likely to be realised with larger-scale processes, the extraordinary energy intensity of nanoproducts, many orders of magnitude above existing traditional materials (Bauer et al. 2008, Kim and Fthenakis, 2012), is likely to be a dominant area of the life cycle impacts. Subtle discrepancies in laboratory measurements could potentially lead to high orders of error when scaled up to larger production levels. As Khanna (2008) concludes, the projected LCA impacts may well be over-estimates when, in all likelihood, process yield and efficiency gains are realised at industrial levels (Khanna, Bakshi et al. 2008).An area presently omitted from many LCA studies is the specific impact attributable to the requirement of high precision instruments and bespoke infrastructure necessary in the formation of materials where precise control and monitoring is required to achieve the desired product. The omission of these elements hamper an accurate ‘full-scale’ estimation of overall life cycle impacts.

The problem associated with scaling issues can be also observed from the variation of LCA results from lignocellulosic ethanol. As most research is still in the early stage of development and has not even reached the pilot scale stage, process simulation is often used to generate data about the industrial-scale process. In such a way, lab-scale data and information from simulation can be used to assess the technology under development. The resulting assumption from process simulation, data generated and predicted scales contribute to the uncertainties of LCA results. Additionally, lignocellulosic biofuel production is anticipated with co-generation of by-products such as electricity and chemicals; the scale of biofuel production with the resulting scale of co-product will affect the choice of selected allocation methods. The results of LCA studies can be prominently influenced due to choices of different allocation method and these may well change as a result of the scale of the operation. For example, if the production of bioethanol from wheat straw is only done as a niche process then the allocation on a mass, energetic or economic basis may be accepted, but if the production of bioethanol becomes the driver for the growth of a field of wheat, it may well be that economic allocation is more commonly chosen. Information regarding the sensitivities resulting from allocation must be reported and shared with all members of the development team to ensure that decisions concerning future direction of development are made appropriately.

Lack of published analysis concerning LCA of novel food processing makes determination of the impact of scaling issues difficult to quantify. Following the rationale proposed by Hospido et al. (2010), the boundary should be drawn such that the analysis concerns only that part of the production chain affected by the change in technique, however in doing this, not only will small discrepancies take on a disproportionate importance, but by neglecting certain elements of the process, full optimisation potential may be prevented due to certain environmentally critical aspects being overlooked. When comparisons are essentially of the changes within versions of the same novel process, e.g. if comparing the impacts of using different component solutions for soaking seeds within the same basic operation, the omission of data concerning equipment that would be required for a commercial facility may not be important, since that omission would be consistent across all comparisons. Difficulties arise however, when comparisons are made against existing, established routes for producing the functionally equivalent foodstuff, for example where process flows and life cycle inventories are developed based on an industrial scale processing facility with all the necessary ancillary equipment. Whilst an LCA can be developed using mass balance and collected energy usages from laboratory test runs, these will not be comparable with industrial scale processing.

Apart from the obvious difference in scale, laboratory production is often completed as a batch process with prominent impacts on energy consumption for start-up and shut down, in addition to potential product wastage through clean-down of equipment. A comparative LCA was performed of the same process – production of food grade oil-bodies, using i) laboratory measurements including energy for start-up ii) laboratory measurements with start-up energy removed and iii) laboratory mass balances projected as a continuous 50 tonne/day production unit, using manufacturers data for equipment energy consumption. Figure 2 shows the results generated, in which it can be seen that even when removing the energy requirements for start-up of the batch production, there is a prominent disparity between the projected industrial scale LCA and that generated using laboratory results alone. Even taking into account the additional energy requirements for material heating and transfer processes at the industrial scale, it is clear to see that basing an LCA on laboratory data alone would give rise to very different conclusions concerning the environmental credentials of the process and potentially lead to ill informed decisions being made. This clearly links in with the requirement to not only identify how the LCA results obtained will be used, but also who will be viewing and using them, both at the time of presentation and in the future.

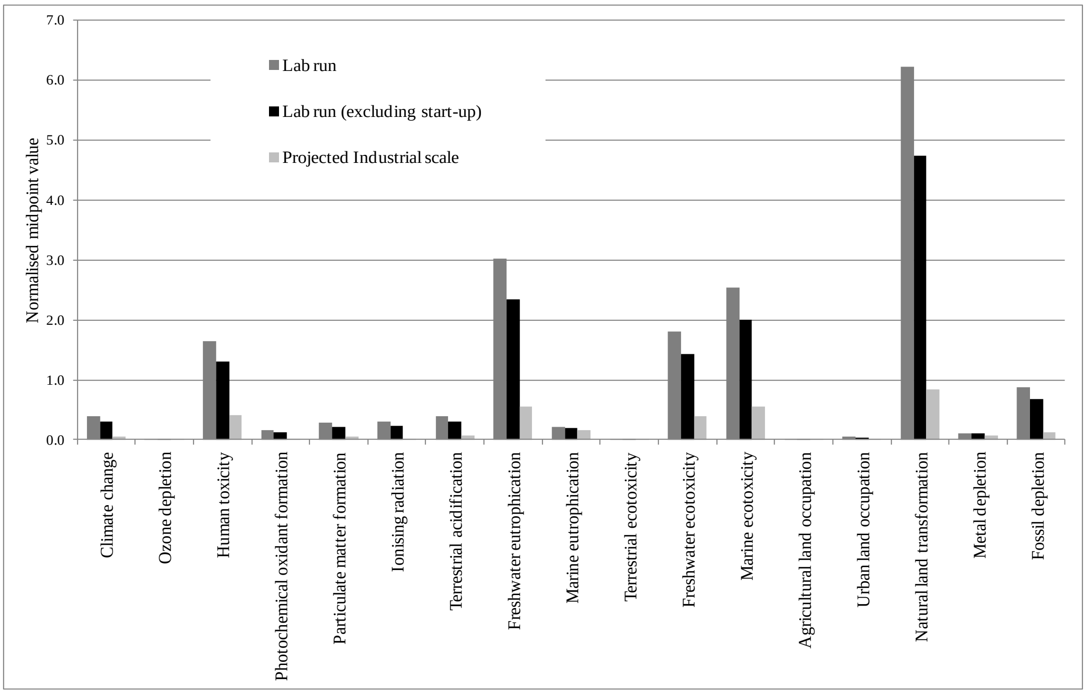

Figure 2: Comparison of oil-body LCA using lab & projected industrial scale data: ReCiPe(2008) midpoint analysis

Other scale up methods could potentially be used for LCA, for example, future scenarios of using new technologies can be estimated by using an economic input and output model to obtain national average data. Process simulation could provide material and energy flows at different scales for LCA, and engineering design could supply infrastructure information, although the use of such design techniques may not always be successful due to lack of data and functionality within the simulation packages for modelling new processes and unusual or novel materials. It must also be noted that the use of these alternative methods for obtaining scale-up data could increase the uncertainty within the process to an even greater extent.

Throughout the three case studies outlined, the range of methods to overcome scaling issues is being investigated by the authors. With the issues outlined here, there could clearly be a case to say that the results from LCAs performed at this early development stage should never be published and that they should be consigned to internal use only. Certainly the results from figure 2 would indicate that a study based purely on laboratory-scale data should never be used to publish a comparative LCA against an existing commercial technology to an external audience. However the authors would argue that publication of information concerning the impact of scale up within novel process LCA is important to be shared within the LCA community and that failure to do so would prevent progress in understanding the complexities and considerations presented by use of LCA at this early development stage. As such, dissemination of LCA results generated using the scaling techniques described here should be encouraged, providing such publications provide clear narrative on the complexities and sensitivities encountered, together with some estimation of uncertainty and adequate caveats on the use of the data. It is anticipated that future research in each case-study will enable the publication of data concerning the uncertainties associated with scaling within LCA to help in the further quantification of this issue.

3.3 Data

For early stage LCA work, speed of assessment is invariably an important factor for providing information at the stage in which changes to the process can most effectively be made, and as noted by Heinzle et al. (1998), to minimise the time to production under patent protection. In this instance, use of secondary data is often the only practical solution, since primary data would either not be available or take too long to gather. There is a wealth of publicly available inventory data for a wide variety of processes and substances. The European Commission Joint Research Council (JRC) publishes a list of available databases, together with its own database of materials, the ELCD database (2012). However, novel processes can often involve the use of new materials or materials that are less prevalent as raw materials within existing processes, furthermore, as noted by Jimenez-Gonzalez et al. (2000) LCA databases represent just a part of the raw materials used in chemical and biochemical companies. The practitioner is thus faced with the dilemma of whether to invest time and resource in primary data collection or to attempt to utilise inventory data for a similar process as a proxy.

Missing datasets for nanomanufacturing processes is a large barrier in conducting valid LCAs. Nanomaterials can be broadly defined as taking particle, fibre or plate forms, however a diverse range of structures and sub-groups stem from these broad categories (ISO 2008; Meyer et al. 2009). Nanomaterials with existing or high potential for future industrial applications are carbon based, composites, metals/alloys, biological, glasses, and ceramics (Bauer et al. 2008). Nanomanufacturing techniques are split: top down; broadly mass change processes and the formation of particles from larger parts, or bottom up; chemical synthesis utilising individual atoms or molecules as the material building blocks ((Ju-Nam and Lead 2008). However, the number of different synthesis routes are continually growing and often unique to the specific nanomaterial formed (Luttge, 2011). It follows, similarly to the assessment of chemicals, that a generic LCA covering all nanomaterials cannot be produced (Klöpffer 2007); the requirement for bespoke nanomaterial datasets is thus required.

In response to missing nanomaterials data, inventory information for bulk material counterparts is often used in place of the actual nanomaterial. Modelling life cycle impacts using bulk materials alone omit downstream life cycle stages required in the production of nanomaterials, which among other factors, such as additional process complexities, have considerable additional energy demands (Bauer et al. 2008; Khanna and Bakshi 2009).

For LCA studies of the lignocellulosic ethanol conversion process, data such as material flow, energy flow, and infrastructure of industrial scale ethanol conversion plant are all needed. Whilst laboratory data could potentially be used to provide some of these, albeit with the issues as previously outlined in 3.2, studies taking into account the manufacturing processes often rely on simulations due to the lack of commercially available data. Together with functional unit and system boundaries, data inconsistencies contribute to the conflicting LCA results of lignocellulosic ethanol in the published literature (Borrion et al. 2012). As most research in the second generation biofuel technology is at laboratory scale, with just a few pilot plant operations, detailed design data is not available in the literature (Searcy and Flynn 2008). Cherubini and Stromman (2011) also highlight the problem with data scarcity of advanced conversion technologies; the few studies that exist are mainly approximations based on mass or energy balances. Furthermore, there is a gap in LCA data for enzyme manufacture, which can vary in its energy input and emission outputs depending on both enzyme family and energy mix at the manufacturing location (Singh et al. 2010). Such data is not available in life cycle databases or published literature (Spatari et al. 2010).

As noted previously, there is a wealth of published literature concerning LCA studies on food ingredients and products, with approximately 40 such papers documented in the abstract and citation database SCOPUS, between 1999 and 2010 (Notarnicola et al. 2012). Despite this the authors have found very little concerning LCAs for novel foodstuffs or processing, with Hospido et al. (2010) and Pardo and Zufia (2012) being two exceptions. Data gathering is one of the issues raised within Hospido et al. (2010) who recommend that specific data should be utilised for the foreground system, whilst average data – with a suitability check, be used for the background system. Within the novel food case covered here, there are several instances where data for previously undocumented materials is required, one of which is the treatment chemical to ensure microbial stability. Similar to the other two case-studies, failure to access such data will necessitate the use of proxy materials to complete the LCA study based on the laboratory scale flow, however if the process were to be commercialised, the activity that requires the proxy data would almost certainly be replaced by a pasteurisation unit.

As shown in each of the case studies discussed here, increasing the coverage of databases and including emerging technologies such as enzymes and nanomaterials is essential for accurate use of LCA within the early stages of research. Where LCI data is not available, the usage and intended audience stipulated within the goal and scope will dictate whether time should be spent attempting to access data for such materials. Such efforts may not be beneficial or sensible where speed of assessment and reporting is required for internal decisions, particularly as the LCA can eventually be updated as more representative data becomes available (Kunnari, 2009) and such data may not be required for commercial scale LCA. However where decisions are to be made based on LCA information generated using proxy materials, some form of sensitivity and uncertainty analysis must ideally be performed to retain the credibility of the model. This is absolutely paramount for communication of results to external parties. As with the previous three focus areas, ensuring that all parties concerned with the development process fully understand the complexities, assumptions and limitations of any data used for the LCA conclusions presented is vital to ensure that decision making is performed appropriately.

3.4 Uncertainty

All LCA studies will have a certain degree of uncertainty and as noted by Heinzle et al. (1998) '*in the design process we can never be sure whether we know all important data and interactions*'. When conducting an LCA it is important to understand how various processes and steps such as goal definition and scoping, inventory analysis and impact assessment impact on the confidence in the results.

Clearly, the issues discussed so far within this paper all contribute towards uncertainty and the integrity of any LCA is dependent on restricting the degrees of uncertainty. Using the analogy of a length of rope to represent a robust LCA study, each degree of uncertainty can be seen as a fray in one of the cords that form the rope. As depicted in Figure 3, where the uncertainty is considerable, the fray becomes a break. When the number of frays is limited, the rope remains intact; however, when there are too many serious frays or break, the rope falls apart. Likewise, with an LCA if the level of uncertainty is too great, the integrity of the LCA is in such doubt that the study becomes at best meaningless and at worst dangerous, as a decision making tool. Given that LCAs on emerging technologies are most often generated to provide information upon which development decisions will be based, whether they be to modify a particular aspect of the process or whether to pursue the development at all, clarity concerning the sources and levels of uncertainty is paramount. The analyst must take absolute responsibility for ensuring that all decision makers are clear about the data provided to them.

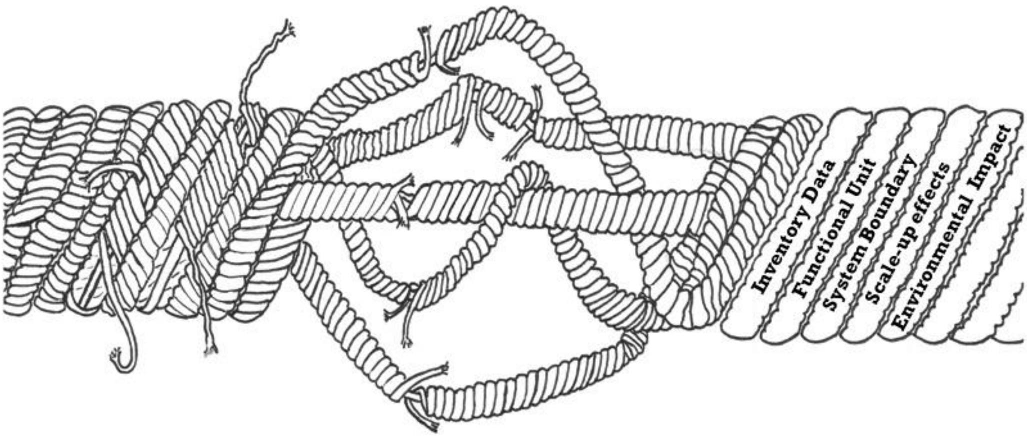

Figure 3 Uncertainty for LCA at early stage

Nanomaterials have only recently begun to be incorporated in mass consumer products and despite touted performance gains, the newness of nanoproducts result in little data in existence for in-situ prolonged usage or disposal (Meyer et al. 2009). Nano-containing goods are subject to degradation with use, with primary effects on functional performance and the matter of released nanomaterials to the environment, the effects of which are of great uncertainty (Oberdörster and Oberdörster, 2005; Som et al. 2010).

Nano-specific end-of-life treatment is presenting challenges for existing waste and recycling practices and strategies (Breggin and Pendergrass 2007; Franco et al. 2007). Additional infrastructure and life cycle stages will foreseeably be required. Wastewater plants have been shown as ineffective in containing certain nanomaterials (Brar et al. 2010) and incineration proposed as a way of precious material retrieval and destruction of potentially harmful materials is facing problems such as the melting temperature of nanomaterials often being higher than bulk material counterparts (Olapiriyakul and Caudill 2009). Incineration can potentially release more thermally stable structures such as carbon nanotubes into the atmosphere (Franco et al. 2007). Recycling of nanomaterials is vital to close the loop and reduce the extraction from finite mineral and metal reserves, to justify the large investment in processing and energy inherent in nanomaterials, and will likely be a mandatory process in the future. However the details are not formulated in any strategy; making the process of conducting LCAs on these emerging technologies all the more uncertain.

A typical LCA study of lignocellulosic biofuel consists of five main stages: biomass production, biomass transportation, biomass conversion to biofuel, biofuel transportation and fuel use in the vehicle. Uncertainties can rise from any of these stages due to data quality, the assumptions made, regional practices and so on. For example, within the biomass production stage, uncertainties can arise from how indirect land use change is accounted for and measured, irrigation practices and fertiliser usage. Within the biomass conversion process, enzyme production, co-generation of different by-products and materials manufacturing can all contribute to a certain degree of uncertainty of an LCA result. In addition, future scenarios such as co-product generation and fuel supply can vary due to the market effect; this will lead to different allocation methods and different application of fuel end use contributing to the uncertainties of an LCA study.

With the creation of LCA studies for novel foodstuffs using alternative techniques, many levels of uncertainty have been encountered. Use of proxy data for seed pre-treatment chemicals, uncertain projections of yield for commercial scale variants of the lab process and changing process requirements all compound the uncertainty that would normally be anticipated within an LCA.

Uncertainty in any LCA is important to quantify and report, however the complexities and timescales involved with analysis of novel processes compound the issue such that the levels of uncertainty are greater and more invasive. Kunnari et al (2009) note that conclusions should be formed (and hence decisions taken) only on the basis of clearly prominent results. Assessment of significance can however be more problematic with the layers of modelling and uncertainty involved with emerging technology assessment and those responsible for delivering the results of such LCAs must ensure that the full details and implications are reported and fully understood by all concerned within the timescale required for decisions being taken.

4. Implications

The growing trend in applying LCA for early stage research can be observed from both outlines of current research projects and within published literature, demonstrating increases in both analysts and audiences for such studies. Clarity of purpose must be paramount for LCAs on emerging technology, with the goal and scope clearly specifying how the results are to be used; whether they are intended to help inform decision makers of environmental "hot spots" and/or to compare the new process routes with current technology. The purpose of the study will affect methodological choices and requirements considerably and those involved with generating novel process LCAs need to ensure that all stakeholders are fully aware of the realities. Practitioners need to be particularly vigilant to the fact that the decision makers within the development cycle are most often not LCA experts and must therefore be fully apprised of the complexities, sensitivities and uncertainties involved, which are far greater than for standard LCA. Whilst speed of analysis and reporting is of the essence, such vigilance in this area is vital to ensure decision making occurs appropriately.

In addition, as the use of LCA becomes more common and required within research, .individuals that are not necessarily LCA experts may well take published material and use it for further study and comparison. Extraordinary care must therefore be taken to ensure that LCA for early research is not underestimated in terms of its complexity within the development cycle and is always performed by suitably qualified individuals.

Table 1 summarises the issues observed within the three case studies, with suggested actions to mitigate the challenges faced. In order to conduct an accurate and meaningful LCA at early research stage, issues such as system boundaries, functional unit, scaling issues, data and uncertainties have to be acknowledged and addressed. Kunnari et al (2009), advocated methodology adjustment to enable LCA to function as a tool for early assessment. From the examples provided here, it is apparent that many of the approaches suggested e.g. scenario analysis, use of proxy data, documentation of uncertainties, can and must be adopted irrespective of the technology under investigation.

Table 1. Summary of main issues in using LCA for early research

Main issues	Challenges	Suggested action for novel LCAs
Comparability	New material not functionally equivalent to that which it replaces The function of the new technology not comprehensively defined Consumption patterns (& thus market conditions) potentially affected by creation of new product	Expand system boundaries to establish functional equivalence wherever possible Depict multiple functional units within studies where necessary, reporting all assumptions concerning future scenarios and technology development Maximise clarity of purpose within goal & scope Report and fully explain all results and sensitivities to decision makers, ensuring full understanding.
Scale	New technology will not entail the same level of complexity at the early stage of development as it will as an industrial scale process Lab-scale results suitable for hot-spot analysis but usage problematic as comparator for large scale New processes may exhibit far lower yield at lab-scale than would be possible in commercial facility	Use process simulation and engineering design to generate data at different scales where applicable Consider estimating future scenarios using economic input/output models to obtain national average data Wherever possible, results from iterative LCAs generated as new processes progress should be published, to build quantitative understanding on how scale-up affects results
Data	Lack of data for new materials Primary data not available or would take too long to gather within development timescale. Data quality reliant on the degree of technology development Environmental impact assessment methodologies will lag behind the formation of new materials with potential impacts in the environment.	Use representative proxy data where necessary to speed analysis, ensuring full details of uncertainties reported and explained to decision making team Provide references and details for data sources and calculation methods as part of novel LCA results. Provide detailed, characterised information regarding material(s) being investigated, to facilitate analysis of the environmental effects within future assessments Encourage publication of work wherever possible and use all data analysis to help in building databases for emerging technologies.
Uncertainty	Unknown future applications Unknown industrial scales Data gaps The degree of technology development Unavailable in-use performance information.	Use estimates of use profile for the intended application, along with projected service life Attempt to assess uncertainty wherever possible Provide transparent information regarding the source of uncertainty, uncertainty level and sensitivities within the novel LCA report and ensure the importance and implications of these are fully understood by all, prior to decisions being taken.

Furthermore, whilst LCAs based entirely on lab-scale data should be limited to internal decision making only, publication of data generated for early stage LCAs and findings from such studies that concern the four areas highlighted within this paper would be beneficial to the growing community of product and process developers and decision makers that wish to utilise LCA to its full effect within the development cycle.

5. Conclusions

This paper highlights the research challenges and issues when applying LCA to early research as illustrated by case studies in three very different sectors, within which the four main areas discussed were comparability, scaling, data accessibility and uncertainty.

Analysis of emerging products and processes intensifies the issues of comparability experienced with LCAs of established systems. Establishing a suitable functional unit and ensuring functional equivalence with current technologies can be more problematic than with standard LCAs, since future applications are not always clear and can be subject to change with the development of new technology. Scalability is one of the most prominent problems when conducting an LCA for early stage. New technology under investigation at the basic concept or lab stage does not entail the same level of complexity as an equivalent industrial scale process and the new processes may exhibit far lower yield than would be possible in a commercial facility. In addition, different processing stages or materials may be required to overcome issues at lab-scale that wouldn't be evident in a commercial facility where they would be redundant or replaced by more 'efficient' alternatives. The resultant early stage LCA may have prominently more variables, complexities and scenarios than a 'traditional' LCA, all of which may have a prominent influence on the results generated and the ensuing assessment of process viability. Those responsible for generating such LCAs must ensure that all parties within the process/product development team are clear on the complexities and sensitivities involved, to ensure decisions are taken appropriately

The reliability of an LCA study at early stage is strongly dependent on the data used. Development of emerging technologies can often use materials that are either novel themselves or infrequently used within industry, with accessibility of inventory data an issue in both cases. Whilst primary data collection may be possible, the time taken for such an exercise is counterproductive to the required expedience for early stage development. Use of proxy data will therefore be more prevalent in such early stage LCA studies, together with the use of data whose quality may not meet the desired level. The authors believe that whilst transparency of data is always important for LCAs, special emphasis should be placed on the reporting, explanation and justification of data within early stage LCA reports such that data can be more easily adapted and augmented as updated information becomes available. In addition, where inventory data is generated pertaining to a material for inclusion within the LCA, that information should be placed in the public domain wherever possible to aid with the development of databases for future use. Published LCA studies reporting detailed inventories and characterised nature of materials are beginning to appear; examples include Griffiths (2013a, 2013b) (Griffiths, O'Byrne *et al.* 2013; Griffiths, Owen *et al.* 2013).

All LCAs have certain degree of uncertainty, and early-stage LCA is not unique in that. However, the source and magnitude of uncertainty increases with such LCA studies due to combined effects described. Failure to acknowledge the uncertainty and fully explore the caveats can result in inefficient use of the information gathered and inappropriate decision making at this key developmental stage. In recognising the difference and uncertainties of LCA within early stage research, development of specific guidance for inclusions within the goal and scope for novel process LCA could be beneficial. These should potentially include the requirement for conclusions to be made only when clearly prominent results are indicated, as suggested by Kunnari (2009), together with more expansive reporting guidelines to ensure all simplifications, projections, sensitivities and uncertainties are not only documented, but adequately conveyed and explained to members of the development team to ensure their full understanding of the issues behind the results presented, before decisions are taken.

Finally, where results are generated from the step-wise improvement in quality of information that inevitably occurs as the technology development progresses, these should wherever possible be compared against initial results and reported within the public domain. This would enable development of a quantified understanding of the order of magnitude difference between early stage results and those generated further down the development cycle.

6. Suggestions for the future

For increased understanding of both the issues concerning the process and the results of LCAs involving emerging technologies it is important that information regarding their execution is published in the public domain. Clearly there may sometimes be issues regarding the reporting of specifics for these projects since by their very nature they may contain sensitive or confidential information, however any information that can assist with the creation and understanding of methodologies for 'novel' LCA studies, even if generalised to protect intellectual property, can only be beneficial. To that end, the authors intend to follow up this article in due course with an update of type and success of strategies used to overcome the challenges discussed here within their practical application. They would also encourage all fellow researchers involved with LCA work on novel processes to publish information beneficial to the development of this area.

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Submitted manuscript

Does Carbon Footprinting Paint The Right Picture For Process Improvements? A Case Study of Mayonnaise Production.

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Abstract

The environmental impacts of rape and sunflowerseed oil mayonnaise were assessed using both Carbon Footprint (CFP) analysis and Life Cycle Assessment (LCA). In addition to identifying the environmental burdens of both systems, a comparison of the results was performed such that the impact of using CFP data alone could be assessed within the wider environmental context of LCA to determine whether process improvements based on the CFP results would be targeted correctly, or potentially cause burden shifting.

The CFPs of 1 tonne of packaged and palletised mayonnaise were found to be 2.2 and 2.4 tonne CO_2e for the rapeseed oil and sunflowerseed oil derived products respectively. In both cases the seed oil provided the largest contribution, with 58% of the sunflower and 62 % of the rapeseed mayonnaise footprint. The next largest contributors were packaging glass and power consumption. Life cycle impact assessment using ReCiPe(2008) revealed that at the endpoint level, the most prominent impact category was agricultural land occupation, followed by fossil depletion, with climate change categories ranking third and fourth. Within these categories, the largest contributors were the seed oils, glass and power. Similarly, climate change is not the most prominent impact category when the normalised midpoint data was reviewed, ranking twelfth out of the seventeen midpoint results, with the toxicity and eutrophication categories showing as having far greater impacts. Midpoint analysis does however confirm seed oil as the largest contributor of impacts.

The comparative analysis demonstrates that within a full LCA, climate change is not indicated as the most prominent environmental burden at either the mid or endpoint stage and therefore an LCA may be a more appropriate assessment tool than CFP. Whether analysed using CFP or LCA however, the process elements highlighted as the largest contributors, thereby having the greatest potential for improvement, are the same. Thus, information from either study would result in impact reduction efforts being targeted consistently. Further assessment is required to determine whether this is a result that would be repeated in all mayonnaise types (or wider) applications.

Keywords: Life Cycle Assessment, Carbon Foot-print, Mayonnaise, Edible Oil, Emulsion

1.0 Introduction

It is widely accepted that urgent action is required to address the causes and consequences of climate change. The UK Government set targets to reduce Greenhouse Gas (GHG) emissions by at least 12.5% by 2012 and 60% by 2050 compared with the baseline emissions of 1990 and this has lead to carbon reduction targets being set throughout industry sectors and individual companies to meet those levels. Within the UK, Food and Drink Federation (FDF) members are committed to an industry-wide absolute target to reduce CO_2 emissions by 35% by 2020 against a 1990 baseline measured within their voluntary Climate Change Agreement with the Department for Energy and Climate Change (DECC) (www.fdf.org.uk (2013)).

Life Cycle Assessment (LCA) has been used widely as a tool to quantify the full range of environmental impacts of systems across the supply chain including food products. Anderson and Ohlsson (1998), Foster et al (2006), Schau and Fet (2008) and Roy et al. (2009) all provide information on the multitude and variety of LCA studies performed in this sector and Notarnicola et al (2012) highlight how important it is that '*we do more integrated LCA studies with regard to our entire food production and consumption system'*. In recent years however, an increased focus on greenhouse gas (GHG) accounting over the entire supply chain has been fostered by such initiatives as the UK 'Carbon Label' and Sweden's 'Klimatmärkning', borne out of a desire to fulfil GHG reduction commitments. This strong focus on carbon reduction has lead to the popularity of the single-issue LCA variant carbon foot-printing soaring. Williams et al (2012) note that carbon footprinting is one of the foremost methods available for helping tackle the threat of climate change through quantifying anthropogenic GHG impact. A search of the bibliographic database 'SCOPUS' on the term 'Carbon Footprint' clearly demonstrates the growth in its usage with an exponential rise in publications on the topic, from 15 in 2000, to 295 in 2008 when the first standard PAS 2050:2008 was published (revised in 2011), to 1061 articles in 2012.

The use of LCA techniques to help drive environmental improvement is to be applauded, however care must be taken to ensure that the use of carbon footprinting does not go against the original rationale for LCA development, to identify the full range of environmental impacts across the life-cycle and thereby prevent burden shifting (www.lct.jrc.ec.europa.eu). As noted by Finkbeiner (2009) climate change is not the only environmental issue of relevance and therefore carbon footprint (CFP) '*is not in all cases the right proxy to support sustainable production and consumption'*, a sentiment shared by many, as illustrated by the favouring of a wider ranging assessment method by countries such as Germany and France and the development of a harmonised methodology for the calculation of the environmental footprint of products (including carbon) by the European Commission (Manfredi et al (2012)).

The purpose of the research presented in this paper is to investigate the relationship between data generated through carbon footprint analysis (CFP) and that generated within a full LCA, to identify whether the information obtained as part of the CFP would enable process improvements to be targeted correctly, or whether the exclusive focus on GHGs would result in environmental impacts, that potentially have a greater significance within the system, being overlooked and burdens being inappropriately shifted.

Using mayonnaise as a case study, environmental performance data has been generated using both CFP and LCA to enable comparisons between the results obtained. The objective was to use CFP to quantify the 'cradle to gate' carbon footprint of both rapeseed and sunflowerseed oil based mayonnaise, then use LCA to identify the relative significance of climate change as an impact category when compared with the wider range of impacts generated by the system. Through analysis of the data generated, the largest process contributors could be identified within each impact category, demonstrating whether CFP and LCA yielded consistent results with regards to the largest contributors within the system and would therefore enable process improvements to be targeted correctly to benefit the environment.

2.0 Method and data

Mayonnaise is probably one of the most widely used condiments in the world today (Depree and Savage, 2001) with its popularity growing enormously since its first commercial production in the early twentieth century. The composition of mayonnaise varies, but has set minimums for oil content with the US Food and Drug Administration (FDA) regulation (21CFR169.140) stipulating a minimum of 65%, whilst the European Federation of Condiment Sauce Industries (FIC) recommends a minimum of 70%. In practice, commercially produced mayonnaise has a fat content of 70 – 80% (Garcia et al (2009)). Despite its high fat content, it is an oil in water emulsion produced using different types of oil, dependant on brand and geographical location, with rapeseed oil and sunflower oil being the most prevalent within Europe.

Mayonnaise can be produced in either batch or continuous processes, with large scale mayonnaise production normally carried out using plant specifically designed for that purpose, which is most often semi-automated (www.edge.silverson.com). Downing (1996) outlines the basic steps for mayonnaise preparation, indicating that the surfactant (egg yolk) is first added to the water and the solution is mixed with an equal volume of oil to form a crude emulsion. This emulsion is then passed through a colloid mill or homogeniser, with more oil being incorporated as required, with the final product being passed on to bottling and further packaging stages.

The objective of this study was to investigate the relationship between data generated through carbon footprint analysis (CFP) and that generated within a full LCA by analysing the cradle to gate production systems for both rapeseed oil and sunflowerseed oil based mayonnaise. The goal was to identify whether environmental performance data generated as a CFP would enable process improvements to be targeted correctly, or potentially cause burden shifting .For the analyses therefore, the Functional Units were set as '1 tonne of rapeseed / sunflowerseed oil mayonnaise produced in UK, packaged in 600g jars, palletised and ready for distribution'. As a cradle to gate study, the starting boundary was the extraction of raw materials, which translated to cultivation of crop and rearing of poultry for the eggs. The finishing boundary was the exit of the mayonnaise packaging facility, thereby excluding use and disposal stages of the life cycle. Attributional LCA models were constructed within SimaPro 7.3.2.for mayonnaise systems incorporating rapeseed and sunflowerseed oil, using the composition as shown in table 1.

Oil	80%	Emulsion formation
Egg yolk	8%	Increase stability
Water	7%	Emulsion formation
Vinegar	3%	Taste, preservation, increase stability
Salt	1%	Taste, increase stability
Sugar	1%	Taste

Table 1: Mayonnaise composition used for LCA,
Source: (adapted from Meeuse et al. 2000),

Inventory data was taken from industry, the ecoinvent database, and peer reviewed literature. A large manufacturer was able to provide details of the steps involved in the production of the mayonnaise, together with some primary consumption data for the foreground process elements. Manufacture and packaging of the mayonnaise was assumed to take place at their manufacturing facility, with data for energy and water consumption at this facility supplied by them. Life cycle inventory (LCI) data for the individual ingredients, packaging materials and energy production were sourced from peer reviewed literature & proprietary databases available within SimaPro, which were discussed and validated with industry. During initial analysis of data it was noted that the results obtained when using different LCI datasets for the cultivation stage of the seed oil generation varied substantially for both oils. Whilst the sources of these differences, which stemmed from differences in input output data for geographical area, were investigated it was noted that in reality, the oil would not be sourced from a single supplier / geographical location, but would more likely be sourced from a range of suppliers / cultivators. To model this most effectively therefore, an aggregate dataset was developed for cultivation for both seeds, assuming that portions came from different locations. The discrepancies smoothed out through using this assumption will be further discussed in section 3, whilst the sources of data for this and the other required LCI datasets are outlined within Table 2.

Seed oil			
Rapeseed cultivation	EcoInvent unit processes, to develop aggregate data set	25% of each of the following: Rape seed conventional, Saxony-Anhalt, at farm/DE Rape seed conventional, Barrois, at farm/FR Rape seed extensive, at farm/CH Rape seed IP, at farm/CH	Nemecek et al (2007) Nemecek et al (2007) Nemecek et al (2007) Nemecek et al (2007)
Sunflowerseed cultivation	EcoInvent unit processes, to develop aggregate data set	50% of each of the following: Sunflower conventional, Castilla-y-Leon, at farm/ES Sunflower IP, at farm/CH	Nemecek et al (2007) Nemecek et al (2007)
Extraction	Industry data amended to UK energy mix*		Unpublished industry data
Refining	Industry data amended to UK energy mix*		Unpublished industry data
Sugar			
Cultivation	EcoInvent unit process	Sugar beets IP, at farm/CH U	Jungbluth et al (2007)
Processing	EcoInvent unit process	Sugar, from sugar beet, at sugar refinery / CH U	Jungbluth et al (2007)
Egg production	LCAFood database	Egg	http://www.lcafood.dk (2013)
Salt manufacture	EcoInvent unit process	Sodium chloride, powder, at plant/ RER U	Althaus et al (2007)
Vinegar manufacture	Based on EcoInvent unit process for acetic acid	6% :Acetic acid, 98% in H20, at plant/RER U 94%: Water, deionised, at plant/CH U	Althaus et al (2007) Althaus et al (2007)
Water (for formulation and general site use)	EcoInvent unit process	Tap water, at user/RER U	Althaus et al (2007)
Packaging glass manufacture	EcoInvent unit process(including 60% recycled material)	Packaging glass, white, at plant/RER U	Hischier (2007)
Packaging film manufacture	EcoInvent unit process	Packaging film, LDPE, at plant/RER U	Hischier (2007)
Packaging board manufacture	EcoInvent unit process	Packaging, corrugated board, mixed fibre, single wall, at plant/ RER U	Hischier (2007)
Euro - Flat Pallet	EcoInvent unit process	EUR-flat pallet / RER U	Kellenberger et al (2007)
Power generation	EcoInvent unit process	Natural Gas, burned in mini CHP plant / CH U	Heck et al (2007)

* - UK energy mix dataset developed using Digest of UK Energy Statistics (DUKES) data combined with EcoInvent unit processes.

Table 2: Details of data inputs for mayonnaise process model

For generation of the carbon footprint, the single issue life cycle impact (LCIA) method IPCC(2007) was used at the 100 year time horizon, whereas for the more complete analysis required for the full LCA, ReCiPe(2008) was used for LCIA with analysis performed at both midpoint and endpoint level. Normalisation of all data was performed using European reference values.

3.0 Results and discussion

The Carbon Footprint (CFP) of mayonnaise produced with sunflowerseed oil was found to be between 2.3 and 2.5 tonnes CO_2equivalent (te CO_2e per tonne of packaged, palletised mayonnaise (FU) dependant on which cultivation dataset was used within the model. The CFP of the sunflowerseed oil mayonnaise using the aggregate seed oil dataset was 2.4 te CO_2e. The Carbon Footprint (CFP) of mayonnaise produced with rape seed oil was found to be between 2.0 and 2.5 te CO_2e per FU, dependant on which cultivation dataset was used within the model, with that developed using the aggregate seed oil dataset being 2.2 te CO_2e.

Analysing the systems developed using aggregate datasets, the breakdown of the CFPs can be seen in figure 1. Here it is evident that the largest single contributor to CFP is the seed oil for both types of mayonnaise, contributing 62.2% of the impacts for the sunflowerseed oil mayonnaise and 58.4% of the impacts for the rapeseed oil mayonnaise. Since the only variable within the two systems was the seed oil, all other contributors had equal amounts of CO_2e irrespective of which seed oil the mayonnaise was based on, although the percentage contributions to the different systems varied, due to the total CFPs of the system being different.

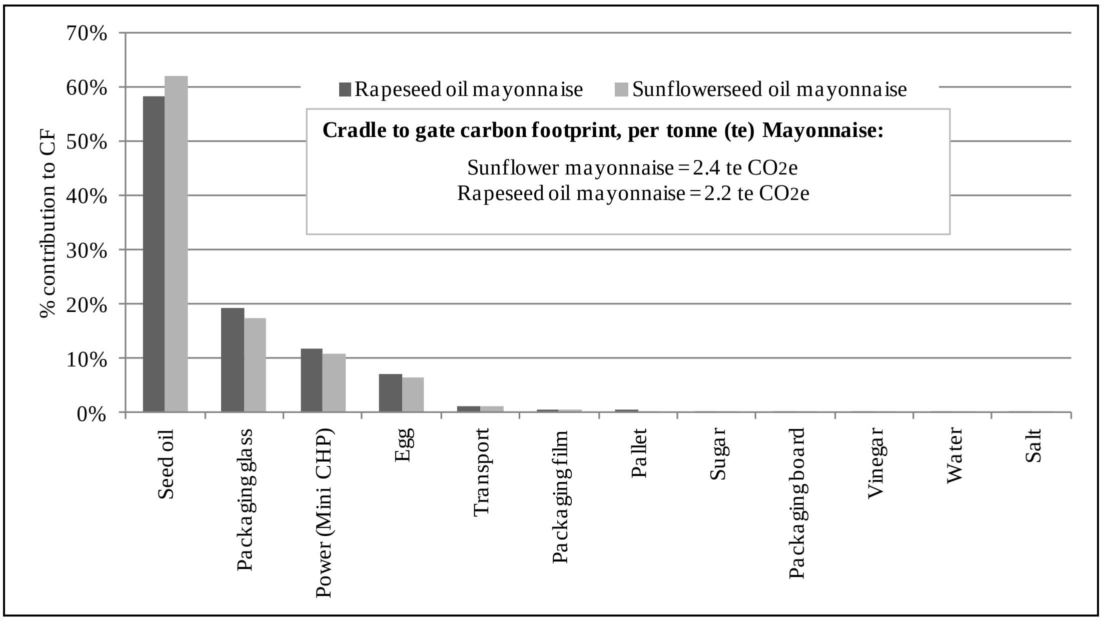

Figure 1: Percentage contributions to carbon footprint of mayonnaise

The second largest impact category for both types of mayonnaise is packaging glass, which yielded a CFP of 0.4 te CO_2e per FU, representing a contribution of 17.5% of the CFP for the sunflowerseed oil mayonnaise and 19.2% for the rapeseed oil mayonnaise. It should be noted that the data for packaging glass takes into account recycling of the glass used as the dataset utilised incorporated 60% recycled glass as part of the glass production process. The third largest contributor to CFP was that of power consumption at the manufacturing plant, contributing 0.3 te CO_2e per tonne of packaged and palletised mayonnaise, representing a 10.9% contribution to the sunflowerseed oil system and a 11.9% contribution to that of rapeseed oil. The power consumption utilised here includes not only the processing power for emulsification, but all energy inputs associated with production of the FU and is an average figure based on production units in different locations.

Drilling deeper into the results for the largest category, that of seed oil data; it was clear that in both cases, the cultivation of the seed provided the greatest contribution to the seed oil CFP, with 87.2% of the sunflowerseed oil CFP coming from the cultivation stage and 82.7% of the rapeseed oil CFP coming from that stage. Shonfield and Dumelin (2005), comment that *'Sunflower oil tends to have high environmental impacts because of the relatively low yields per hectare compared to other crops'* and it is this higher impact that causes the CFP of Sunflowerseed oil derived mayonnaise to be higher than that of its rapeseed oil counterpart.

Looking at the midpoint results for the full LCA, climate change does not feature as the most important impact for either of the types of mayonnaise; in fact when reviewing normalised midpoint data shown in figure 2, it is twelfth out of the eighteen midpoints. For rapeseed oil mayonnaise, terrestrial eco-toxicity is indicated as the most prominent impact category, with a normalised value of 8.1 (compared with 0.2 for climate change) within which 99.8% of the impacts are derived from the use of pesticides for the cultivation of seed for the oil used. This differs from the sunflowerseed oil mayonnaise, where the most prominent impact is calculated as being marine eutrophication, with a normalised value of 5.3 (compared with 0.2 for climate change); however this again is derived principally from the cultivation of the seed, with 96.0% arising from the cultivation stage, through the use of herbicides.

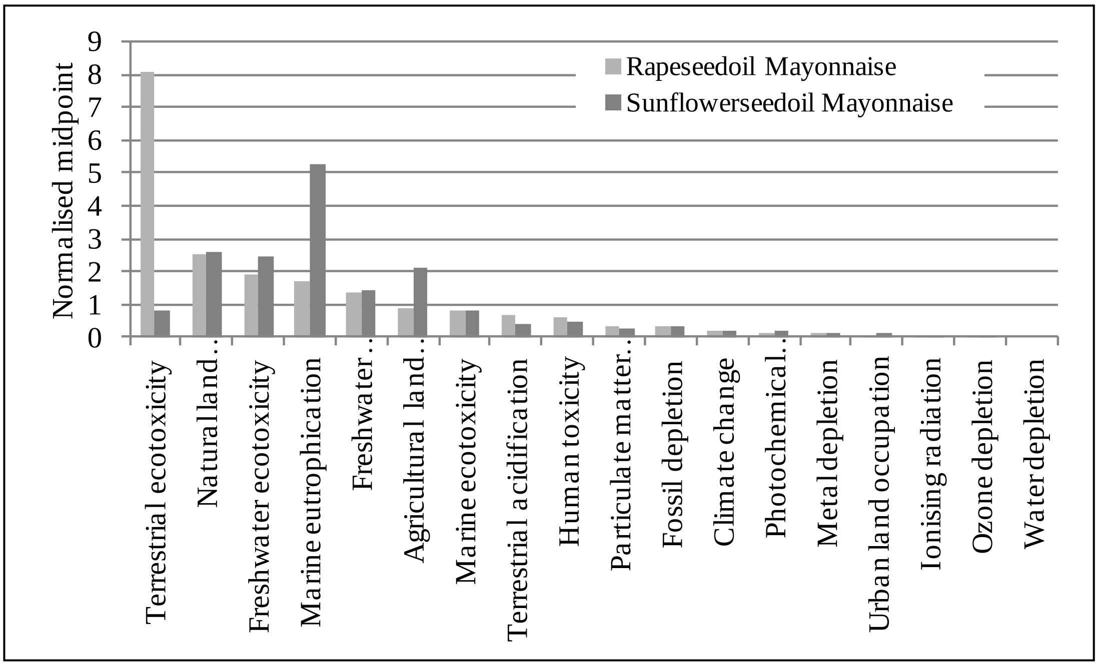

Figure 2: Normalised midpoint analysis for both types of mayonnaise

Whereas the next largest contributors within the CFP were packaging glass and power consumption at the manufacturer, these process inputs are considerably less important for the contribution to the categories of terrestrial eco-toxicity and marine eutrophication, where the second largest contribution to the calculated impact is supplied by the egg production and associated processes. As indicated previously however, the impact from the seed-cultivation stage is so dominating within these categories that secondary contributions are minor; in this case 3.7% for the sunflowerseed oil mayonnaise and a negligible contribution for the rapeseed oil mayonnaise.

It is clear from this analysis that whilst climate change does not feature as a prominent impact when compared against the wider range of normalised impacts assessed through the full LCA, the results generated indicate that the vast majority of impacts within those categories which do feature as prominent, arise from the seed-cultivation stage through the use of agro-chemicals. From this perspective therefore, the results of both CFP and LCA yield consistent results with regards to the areas of production that should be targeted to gain the greatest improvements in environmental performance.

Whilst normalisation of impact values is designed to facilitate the identification of contributors to the environmental burden of a product system, it is also subject to an element of uncertainty brought about through incompleteness in lack of emission data, characterisation factors, or both (Heijungs et al., 2007), Van Hoof et al (2013) state that ranking based on normalised indicators can vary considerably based on the approach used and highlight midpoints that have a lot of contributing elementary flows (e.g. toxicity indicators) as potentially having a higher level of uncertainty through incompleteness in characterisation factors. They go on to advocate combining the use of normalised endpoint indicators for ranking of significance with characterised midpoint values for reporting results. Analysis was therefore also performed using the endpoint category indicators within ReCiPe, again using the heirarchist view and European normalisation to identify the significance of climate change within the normalised endpoint results.

The endpoint results for mayonnaise produced with both oils, as depicted in figure 3, are broadly the same, with agricultural land occupation being calculated as the most prominent normalised endpoint, followed by fossil depletion. Within the endpoints, climate change is separated into two impacts, representing the impact of climate change on human health and the impact of climate change on eco-systems; whilst the endpoint results indicate that, as with the midpoint analysis, the largest impact category is not climate change, the two climate change categories feature as third and fourth among the seventeen indicators.

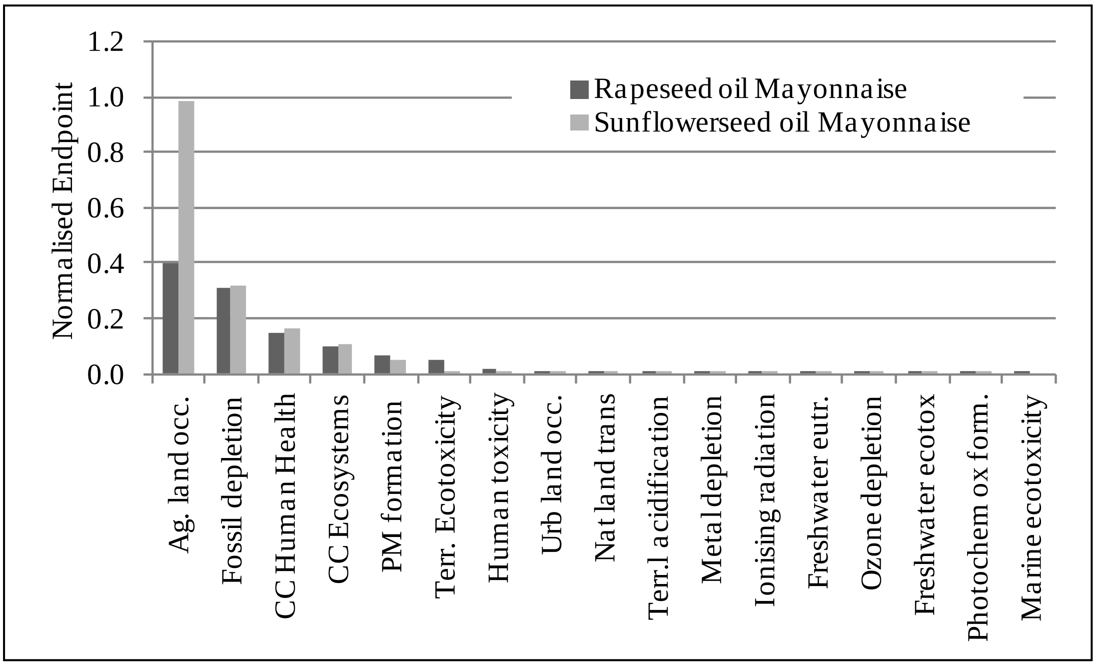

Figure 3: Normalised endpoint analysis for both types of mayonnaise

The actual result for agricultural land occupation is very different for the two systems, however, with much larger results being evident for the system comprising sunflowerseed oil. Cultivation of the rapeseed provides 91.5% of the impact results in this category and Sunflowerseed provides 96.8%. With such a large contribution from cultivation, the larger acreage required for producing sunflowerseed compared with that of rapeseed, as highlighted in Shonfield and Dumelin (2005) is clearly shown within the results here.

The relative contributions to the fossil depletion endpoint category are far less skewed to the seed oil component. Whilst the oils are still the largest process contributors within this category at 44.0% and 43.4% for rape and sunflower based mayo respectively, prominent contributions are also made from packaging glass, where the contribution is 29.3% for the rape and 28.6% for sunflowerseed oil variants. Power consumption at manufacturer also yields a prominent portion of the impacts at 20% for the rapeseed oil based product and 19% for the sunflowerseed oil variant. The order of process contribution within this category is the same as that evaluated for the CFP, indicating that based on endpoint, assessment of the system contributions beyond that of cultivation would yield consistent results with regards to relative process contributors, whether assessed using CFP or endpoint LCA techniques.

Within both climate change impact categories, the seed oil again provides the largest contribution at 59.1% and 62.2% for rape and sunflower based mayo respectively, however as with the contributions to the fossil depletion endpoint category, the second largest process contributions are made by packaging glass and power consumption, at 19.5% and 12.1% for the rapeseed oil system and 17.5% and 10.9% for the sunflowerseed oil system. This further clarifies that for the mayonnaise case study examined here, the endpoint results from the full LCA support the overall findings of the CFP with regards to the relative contribution and therefore importance of the most prominent stages within the overall process.

5.0 Conclusions

The purpose of the research presented here was to evaluate the extent to which the generation of carbon footprint data for mayonnaise produced with either rapeseed oil or sunflowerseed oil, would provide consistent results when compared with impacts identified using a full LCA. Having performed analysis using both midpoints and endpoints, it is clear that climate change does not feature as the most prominent impact category with either measure, with it presenting as third and fourth in endpoint analysis, and twelfth out of seventeen impact categories within the midpoint analysis. From that raw information it could therefore be said that to focus attention on GHG through the generation of and reliance on CFP data would have the potential to cause other prominent areas of environmental impact to be missed and potentially result in decisions being taken that may not have the best environmental outcome.

Further detailed analysis of the LCA results however demonstrate that whilst climate change is not the most prominent impact category, it is the cultivation stage that contributes greatest to the climate change impact category and is also highlighted as most prominent within the wider LCA results. Focussing attention on the greatest impact generator as determined through CFP would therefore lead to activities that would support impact reduction in the areas highlighted as most prominent within the wider LCA. In other words, whilst climate change is not highlighted as most important within the full LCA, the actions that would be required based on the results of the CFP alone would be beneficial to other impact categories that are highlighted as important within the overall full LCA.

Although the impact from the seed oils is by far the largest contributor, should the recipient of CFP data wish to target areas beyond the seed oil, the CFP would highlight that packaging glass and power consumption would be the next most prominent areas to target for impact reduction. These findings would be supported by the data from endpoint analysis within the full LCA where packaging glass and power were both prominent contributors to the fossil depletion category, which was the second most prominent category when reviewing normalised data. Thus again, whilst not directly focussing on the most prominent impacts, any actions taken as a result of data from the CFP would be beneficial to the most prominent impacts identified through full LCA.

It is clear, therefore, that when analysing the mayonnaise cradle to gate production system, carbon footprint and full LCA data yields consistent results with regards to the areas most beneficial for targeting to reduce environmental impacts. In this instance, the use of the single issue LCA variant would not lead to burden shifting within the system.

6.0 Suggestions for the future

LCA was designed as tool to enable the full environmental load of a system to be identified across multiple impact categories and thereby prevent burden shifting by over-reliance on a single issue. The results presented within this paper demonstrate that for this particular system, the actions that would be taken based on either CFP or LCA would provide consistent benefit across multiple impact categories. This may not always be the case however, and in some instances, within particular industrial systems, focus on CFP / GHG reduction may have a negative overall effect across the wider environmental impact categories by requiring steps to be taken that would increase the system impact within other areas of the environment, The authors suggest that publication of more comparative studies of this kind would be beneficial to aid understanding of the benefits or issues of using carbon footprinting as the preferred environmental measure and the focus on GHG reduction targets.

Provision of more data such as this within the public domain would build understanding on consistency of single issue analyses versus full LCA and as such, it is suggested that where CFP results are published, for example within journal or conference publications, these should wherever possible be accompanied by some form of additional analysis to outline the significance of climate change within the breadth of environmental impacts.

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Paper presented orally at conference

Comparison of allocation and impact assessment methodologies on the life cycle assessment of rape and sunflowerseed oils.

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Abstract

As part of the DEFRA Funded Sustainable Emulsion Ingredients through Bio-Innovation (SEIBI) project, attributional Life Cycle Assessment (LCA) models of both sunflower and rapeseed oils have been developed to enable the relative environmental burdens within both production systems to be identified, from cultivation through to factory gate, using existing technologies. This paper shows the effect of using two different methodologies for both co-product allocation and life cycle impact assessment (LCIA) when modelling the life cycle inventories of the two product systems. Results obtained showed that changing allocation methodology prominently changed both the relative contributions of the individual process stages and the relative contributions from the impact categories. This change was heightened when changing both LCIA methodologies.

1. Introduction

The SEIBI project (Sustainable Emulsion Ingredients through Bio-Innovation) is a DEFRA funded collaborative and cross disciplinary project incorporating researchers from the Universities of Nottingham and Bath together with a consortium of industrial partners. The project was initiated to investigate novel processing routes for the production of edible oil emulsions for food production, since a prominent proportion of edible oils are consumed as emulsions, in products ranging from sauces and drinks to confectionery and spreads.

Current oilseed processing techniques involve extraction and refining of the oil using high temperatures and organic solvents, followed by re-encapsulation of the oil if required using manufactured surfactants, for incorporation into a range of food products. The SEIBI project aims to reduce the number and complexity of processing steps required for this process, with the intention that a simplified process will improve efficiency and reduce the environmental impact of the production.

Life Cycle Assessment (LCA) has been used to build models of both the rape and sunflowerseed oil systems in order to i) identify and quantify the relative contributions of each processing step, such that process improvements can be targeted to specific areas and ii) to identify the current environmental loads to be used as a comparison with those generated by the novel process.

Both processes involve the production of not only the product of interest, but also a co-product during both the extraction (meal) and refining (acid-oil co-product) stages. Product systems such as this require the issue of allocation to be considered, to determine the proportion of the environmental impacts that will be attributed to the production of each product. Allocation is defined as *'partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems*' [1]. ISO 14044:2006 [2] states that allocation should be avoided where possible, in favour of system expansion (and the subsequent development of a consequential LCA), however as highlighted in the U.S EPA Guidance document [3], expansion of systems is not possible in all cases and it can be argued that choice of allocation method should be based on what type of LCA is being done [4].

ISO 14044:2006 [2] goes on to state that where allocation cannot be avoided, it should be done is such a way as to reflect the physical relationships between the co-products, although in step 3 of its allocation procedure it acknowledges that this is not always practical [6]. Whilst use of mass as the allocation basis appears to be the preferred approach [5], other methods such as economic value, energy content, volume or even nutritional value (for foodstuffs) can also be used. [5, 6, 7]. From studying a range of published oilseed LCAs it was evident that for rape and sunflowerseed oils, the favoured allocation method is generally economic. The basis for this is that oil crops are harvested for their oil, without which, they would not be financially viable to grow (the exception is soy bean oil – which is primarily grown for animal feed from the meal).

The choice of allocation approach can have a profound effect on the results generated [5,8.9] and it is this effect that is examined with specific reference to the rapeseed and sunflowerseed oil LCAs within this paper.

Life cycle impact assessment (LCIA) aims to provide additional information to help assess the results of the life cycle inventory (LCI), such that the environmental loads can be better understood [7,2]. There are many LCIA methodologies and the one chosen is largely dictated by the impact categories required within the scope of the study.

This paper will furthermore examine the impact that LCIA methodology has on the effect of differing allocation parameters.

2. Methodology

Attributional LCA models of both product systems were constructed using the SimaPro 7 software system.

Functional unit and system boundaries

The functional units of both systems were set as ‘receipt of 1 ton of refined oil at food processor’ with the system boundary starting at the cultivation stage and finishing at delivery of oil to food processor. The process flow used for both product systems was as depicted in figure 1, with the main process stages being cultivation, extraction and refining. For analysis purposes, transportation was aggretated to form a fourth 'process' step.

Whilst this indicates a relatively simple flow-sheet, creation of the LCA entailed each input being further expanded to include the mass and energy balance around each individual system. The result was a complex process network involving over 2000 process nodes (input values).

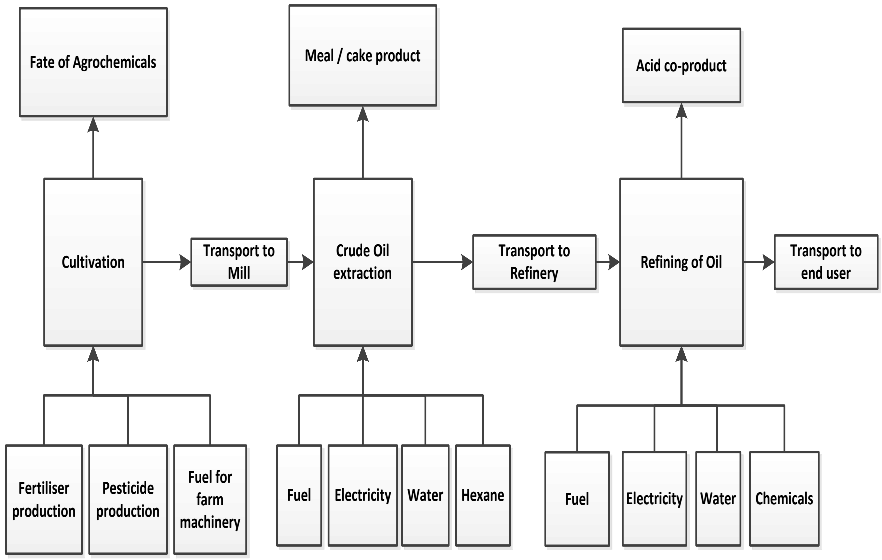

Figure 1. Process flowchart for oilseed processing system.

Data and sources

Data for all stages of the production sequence was taken from Unilever manufacturing sites and suppliers [10,11] corroborated against data from literature sources [12,13,14,15]. Data for secondary processes such as electricity and steam generation was taken from the EcoInvent database supplied within SimaPro, for the geographical area of the process in question e.g. for generation of electricity used in Rapeseed oil extraction, the Germany power mix was utilised. Table 1 depicts the geographical specificity of the data requirements.

Table.1: Geographical scope of LCI data

	Rapeseed oil	Sunflowerseed oil
Cultivation	Germany	South Africa
Extraction	Germany	South Africa
Refining	Netherlands	Netherlands
Transport farm to mill	Road 65km	Road 100 km
Transport mill to refiner	Road 650 km	Sea 12300 km Road 20 km
Transport refiner to factory	Road 50 km	50 km

Allocation methodologies

Allocation was performed using both mass and economic methodologies to facilitate a comparison of results.

The economic allocation was based on market prices [16], combined with the mass balance figures and entailed that within the extraction stage, 76.9% and 82.4% of the impacts were allocated to Rapeseed Oil and Sunflowerseed Oil respectively, rather than their meal co-products. When this was changed to mass allocation, the oils both had the reduced figure of 40% allocated to them. Within the refining stage, economic allocation attributed both oils with 66.67% of the load, whereas mass allocation increased this to 96.45%.

Impact assessment methodologies

Two LCIA methodologies were chosen for assessment of the system to illustrate the differences that can arise through choice of LCIA method. Eco-Indicator 99 (EI-99) [17] is an endpoint method developed by Pré consultants to supersede their Eco-Indicator 95 method. Within EI-99, the results of the LCI are characterised into 11 impact categories and then aggregated into three damage categories namely 'Human Health', Ecosystem quality' and 'Resources'.

ReCiPe 2008 [18] was developed through a collaboration with Radboud University Nijmegen, CML and Pré which was aimed at harmonising the CML midpoint and Pré endpoint methodologies. As such, ReCiPe has 18 midpoint categories and 17 endpoint categories which, like EI-99 can be aggregated into 3 damage categories; Human Health, Ecosystems and resources.

For ease of comparison, the endpoint data only is reviewed in this paper.

3. Results

The inventories were analysed with the objective of identifying both the relative contributions from each of the process stages and the overall environmental load of both systems. The effect of using the different allocation and impact assessment methodlogies was scrutinised on that basis.

Relative contribution from process stages

From the data presented in figures 2 to 5, it is evident that for both oilseed systems, cultivation contributes the largest impacts within each damage category, regardless of allocation or impact assessment methodology used. However when changing from mass to economic allocation, the environmental burdens of each process stage change prominently. When analysed using ReCiPe 2008, increases of 25% and 30% arise for Rapeseed cultivation and extraction, and decreases of 45% and 27% for refining and transport. This change also takes place within the Sunflowerseed system where the environmental loads attributed to cultivation and extraction both increase by 30%, with the loads from refining and transport decreasing by 45% and 26% respectively.

Prominently, these changes lead to a modified order of relative contribution within the system. Both systems retain cultivation and transport as the stages with the greatest contribution, but moving from mass to economic allocation reverses the order of the remaining two, with extraction having a reduced environmental load compared to refining.

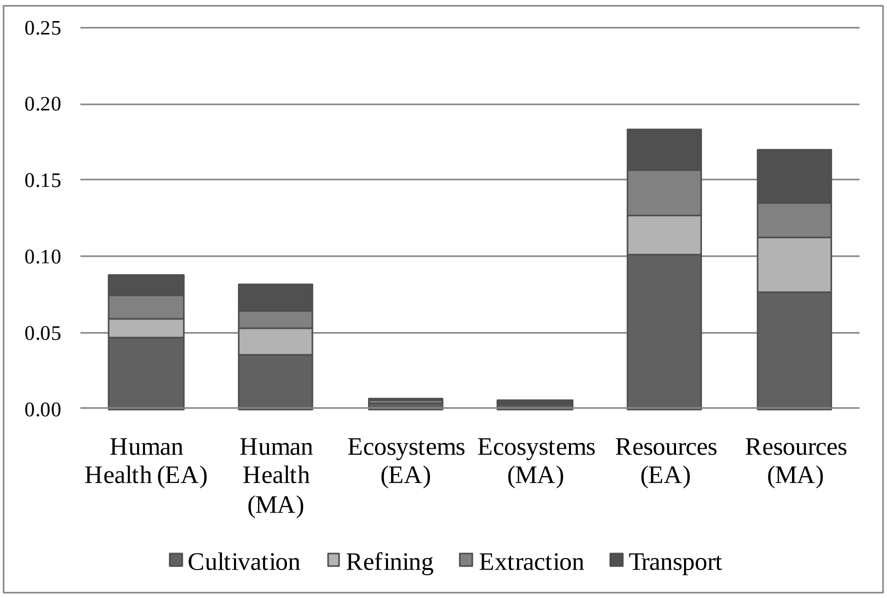

Fig.2: Changes in relative contributions for normalised endpoint data (rapeseed oil system, ReCiPe LCIA methodology)

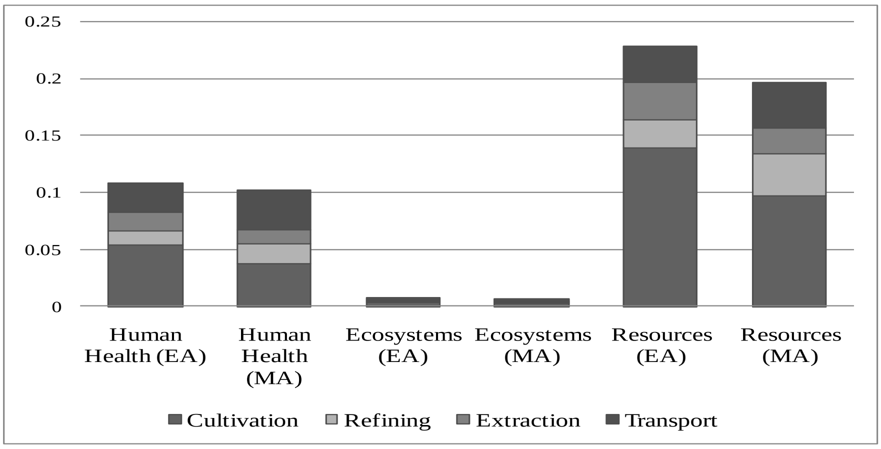

Fig.3: Changes in relative contributions for normalised endpoint data (sunflowerseed oil system, ReCiPe LCIA methodology)

This same effect is observed when using EI-99 as the LCIA method, as shown in figures 4 and 5. Here, within the Rapeseed oil system a move from mass to economic allocation produces increases of 25% and 25% for the cultivation and extraction stages, and decreases of 45% and 31% for the refining and transport stages. Again, for the Sunflowerseed oil system, the same change causes an increase in the environmental load of 30% from both the cultivation and extraction stages, with decreases of 45% for cultivation and 26% for transport. As previously, the order of relative contribution within the system is changed, with cultivation and transport being the largest two regardless of allocation, but extraction moving to third when economic allocation is applied.

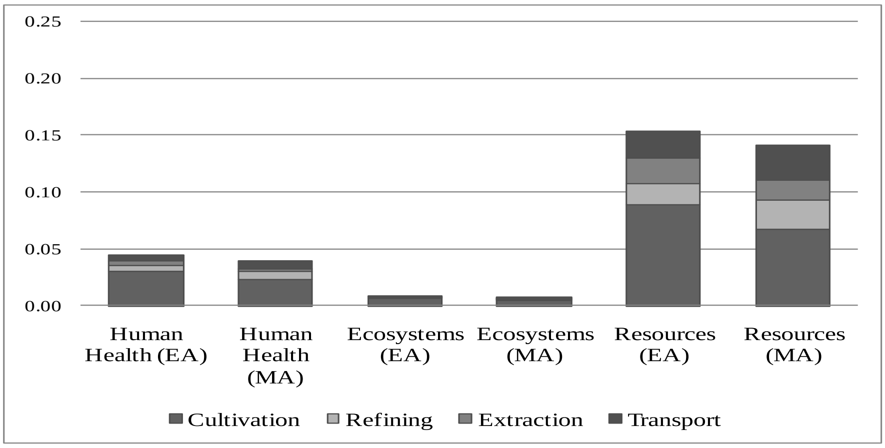

Fig.4: Changes in relative contributions for normalised endpoint data (rapeseed oil system, EI-99 LCIA methodology)

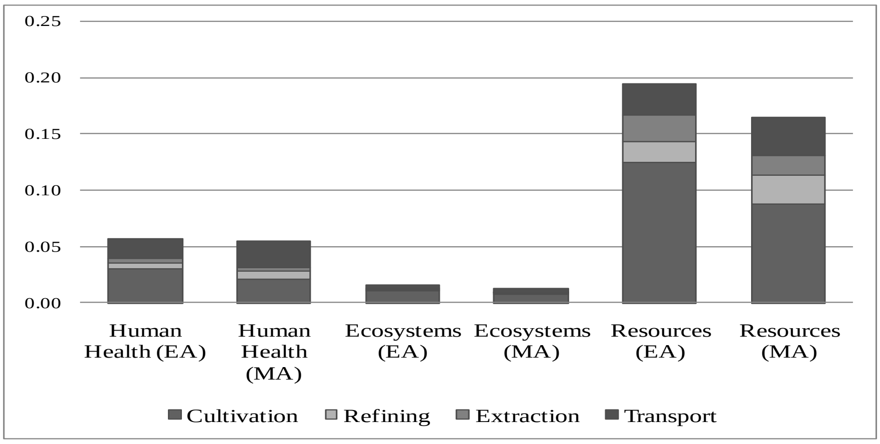

Fig.5: Changes in relative contributions for normalised endpoint data (sunflowerseed oil system, EI-99 LCIA methodology)

Relative contribution from impact categories

Both oilseed systems were analysed to ascertain which of the impact categories were most prominent, and whether that was affected by the use of allocation method. Figure 6 shows the percentage change to characterised impact categories that arise from a change from mass to economic allocation, when usng ReCiPe 2008 as the LCIA method.

For the Rapeseed oil system, within certain impact categories, the change in allocation method has a large effect, with changes of over 20% occurring. However, when the percentage changes are shown for the 4 impact categories that have the largest impact (when comparing normalised data) these changes whilst still prominent are more modest, at 5.8% for 'Climate Change Human Health', 8.2% for 'Human toxicity', 8.7% for 'Particulate Matter Formation' and 7.2% for 'Fossil depletion'.

The changes are more striking within the Sunflowerseed oil system, where both positive and negative changes are found. Here the top 4 impact categories have relative changes of 9.9% for 'Climate Change Human Health', 11.6% for 'Human toxicity', -5.4% for 'Particulate Matter Formation' and 13.7% for 'Fossil depletion'.

Despite these changes however, the relative contribution of the impact categories remain unchanged and allocation method does not affect the order, with Fossil Fuels being the largest contributing category, followed by Respiratory Organics, Carcinogens and Climate Change.

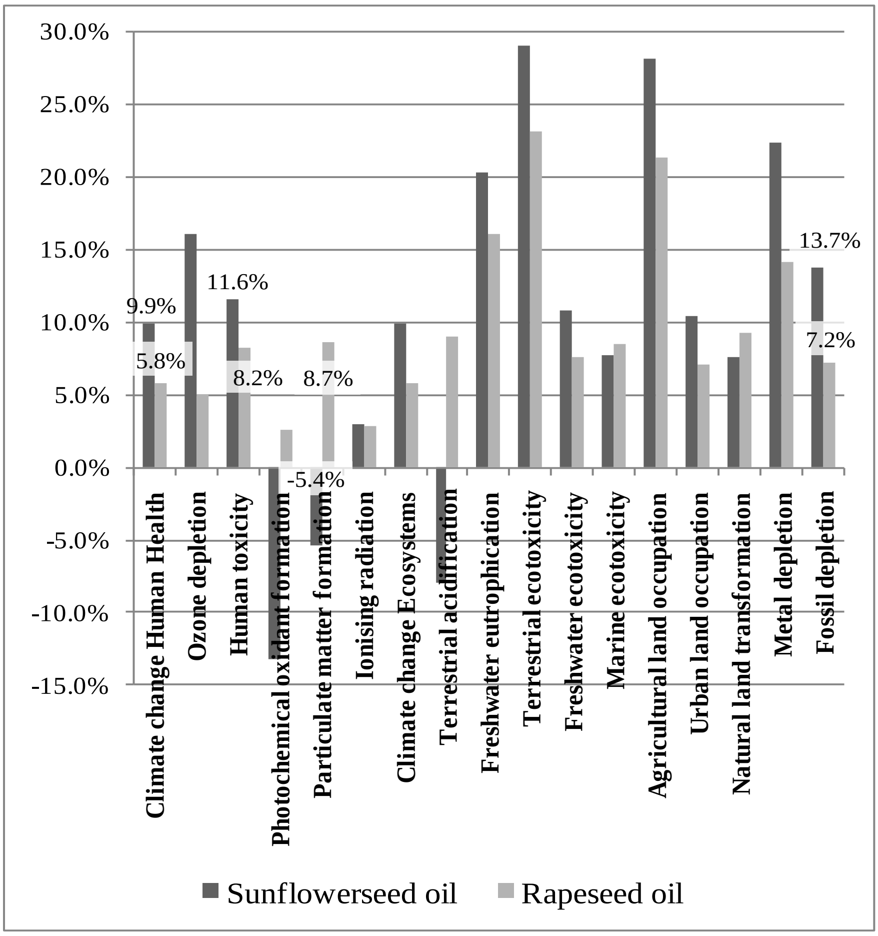

Fig.6: Percentage change in characterised impact category through changing from mass to economic allocation (ReCiPe LCIA methodology)

The changes are even more prominent when using the EI-99 method. Here in terms of percentage change in impact category, there are large changes up to 26% for the sunflowerseed oil systems and 18% for the rapeseed oil system as depicted in figure 7. When the top 4 categories are scrutinised however, the percentage changes are again more modest, but higher than those observed when using ReCiPe.

For the rapeseed oil system, the 'Carcinogens' category has the largest change of the four, at 16.8%, followed by 'Respiratory inorganics' at 9.0%, 'Fossil fuels' at 7.8% and 'Climate change' at 5.7%. Within the sunflowerseed oil system, of the 4 most prominent categories, 'Carcinogens' has the highest relative change at 18.2%, with Fossil fuels at 14.9%, 'Climate change' at 9.9% and 'Respiratory inorganics' at -2.6%.

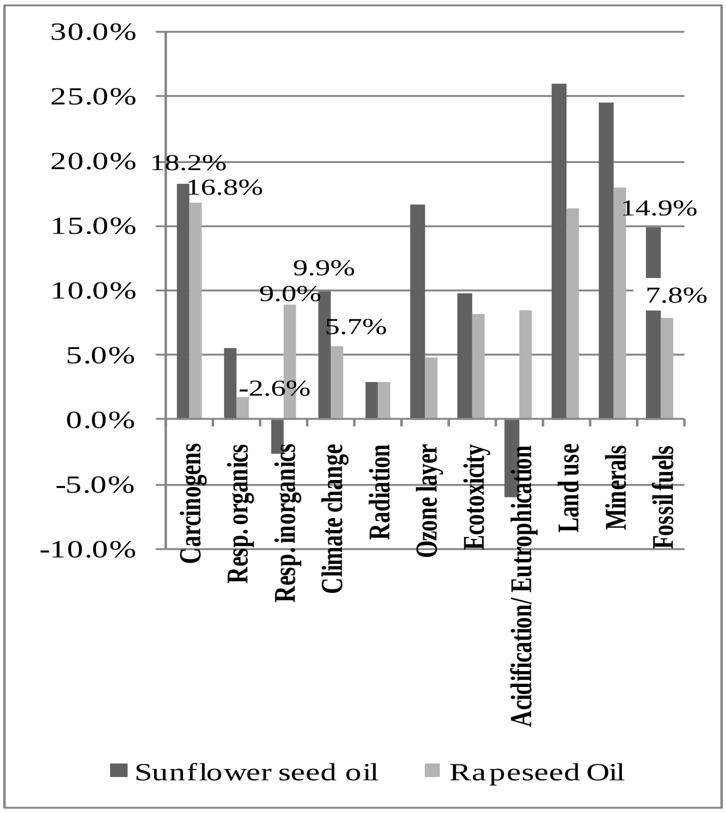

Fig.7: **Percentage change in impact category through changing from mass to economic allocation (EI-99 LCIA methodology)**

4. Conclusions

Where system expansion is not possible or feasible, allocation of environmental impacts must take place within multi-output systems. Several sources acknowledge that choice of allocation approach can have a prominent effect on the results generated [5,8.9]; this is consistent with our findings, reported in this paper, from the analysis of the rapeseed oil and sunflowerseed oil systems, using both mass and economic allocation.

The results of the paper have further shown that the size of the effect is also affected by choice of LCIA methodology, with the relative changes to environmental impact categories for the four most prominent impact categories (based on normalised endpoint data), being greater when Eco-Indicator 99 was used, rather than ReCiPe 2008.

One of the purposes of the LCA determination within the SEIBI project is to identify and quantify the relative contributions of each processing step, such that process improvements can be targeted to specific areas. It is evident from the results presented here, that choice of allocation parameter will be an important consideration for this project (and others with similar scopes) and must certainly be transparent to enable effective decisions to be made based on the LCA results.

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Hetherington A C, McManus M C, Gray D A. (2012). Carbon Foot-print Analysis and Life Cycle Assessment of Mayonnaise production. A comparison of their results and messages: *SETAC Europe 18th LCA Case Study Symposium, 4th NorLCA Symposium, November 26-28 2012, pp48. Copenhagen, Denmark.* Abstract.

Paper presented orally at symposium

Carbon Foot-print Analysis and Life Cycle Assessment of Mayonnaise production. A comparison of their results and messages.

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A comparison of Carbon Foot-print analysis (CFA) and Life Cycle Assessment (LCA) results was performed for the mayonnaise production system. The purpose of the analysis was to ascertain both the carbon footprint and wider environmental burdens thereby identifying whether the methodologies provide a consistent message concerning hotspots within the system and determine to what extent a full LCA would provide enhanced understanding.

Mayonnaise is an oil in water emulsion containing approximately 70 – 80% fat (Depree and Savage (2001)) .The type of oil used varies according to brand, with Kraft using predominantly soybean oil (www.kraftrecipes.com) and Hellmann's using rapeseed oil (www.hellmanns.co.uk). This paper reports on the cradle to gate CFA and LCA performed using a rapeseed oil based mayonnaise as the end product, with the functional unit of '1 tonne of rapeseed oil mayonnaise produced in UK, packaged in 600g jars, palletised and ready for distribution'..

Analysis was performed in accordance with PAS2050:2011and ISO 14040:2006 using SimaPro 7.3.2 with Life cycle impact assessment (LCIA) performed using IPCC (2007) and ReCiPe (2008) at both mid and endpoint levels.

The carbon footprint of 1 tonne of packaged & palletised Mayonnaise was found to be 1.95 tonne CO_2e, with rapeseed oil providing the largest contribution, with 53.89 % of the footprint. The next largest contributors were packaging glass and power consumption, with 21.43 % and 13.31% of the footprint respectively.

LCIA using ReCiPe(2008) revealed that at the endpoint level, the largest contributors to single score were again rapeseed oil, glass and power although the most prominent impact category for each of these was fossil depletion, rather than climate change, which ranked second. Similarly, climate change is not the most prominent impact category when reviewing the normalised midpoint data, with the toxicity and eutrophication categories having far greater impacts. Midpoint analysis does however confirm rapeseed oil as the largest relative contributor.

The analysis performed shows that climate change is not indicated as the most prominent environmental burden at either the mid or endpoint stage when the full spectrum of environmental impacts are analysed within an LCA. However whether analysed using CFA or LCA, the contribution from rapeseed oil is highlighted as the most prominent within the process. Thus, information from either study would result in impact reduction efforts being targeted consistently.

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KEYWORDS

Mayonnaise, Carbon Foot-print, Life cycle Assessment, Rapeseed oil

APPENDIX B. **DATA AND SUPPORTING INFORMATION**

Chapter 4 - Supplementary material

Table B4-1: Allocation factors for use in seed oil LCA models

	Rapeseed extraction	**Sunflowerseed extraction**	**Rapeseed refining**	**Sunflowerseed refining**
Oil price	1151.53	1284.27	1151.53	1284.27
Oil quantity	1000	1000	1000	1000
Meal price	283.77	283.77	575.765	642.135
Meal quantity	1500	1500	36.85	36.85
SEED OIL LCA				
Economic allocation				
Oil - extraction	0.7301	0.7511	0.9819	0.9819
Meal - extraction	0.2699	0.2489	0.0181	0.0181
Mass allocation				
Oil - extraction	0.4	0.4	0.96446	0.96446
Meal - extraction	0.6	0.6	0.03554	0.03554

Table B4-2: Overview of choices for three perspectives - ReCiPe LCIA methodology.

To midpoint impact category	Perspectives I	H	E
Climate change	20 yr time horizon	100 yr	500 yr
Ozone depletion	-	-	-
Terrestrial acidification	20 yr time horizon	100 yr	500 yr
Freshwater eutrophication	-	-	-
Marine Eutrophication	-	-	-
Human toxicity	100 yr time horizon Organics: all exposure routes Metals: drinking water and air only carcinogenic chemicals with TD_{50} classified as 1,2A, 2B, by IARC	Infinite All exposure routes for all chemicals all carcinogenic chemicals with reported TD_{50}	Infinite All exposure routes for all chemicals all carcinogenic chemicals with reported TD_{50}
Photchemical oxidant formation	-	-	-
Particulate matter formation	-	-	-
Terrestrial eco-toxocity	100 yr time horizon	Infinite	Infinite
Freshwater eco-toxocity	100 yr time horizon	Infinite	Infinite
Marine eco-toxocity	100 yr time horizon Sea+ocean for organics and non-essential metals. For essential metals the sea compartment is included only, excluding the oceanic compartments	Infinite Sea+ocean for all cehmicals	Infinite Sea+ocean for all cehmicals
Ionising radiation	100 yr time horizon	100,000 yr	100,000 yr
Agricultural land occupation	-	-	-
Urban land occuptaion	-	-	-
Natural land transformation	-	-	-
Water depletion	-	-	-
Mineral resource depletion	-	-	-
Fossil fuel depletion	-	-	-

Source: Goedkoop et al. (2013).

Table B4-3: The quantitative connection between midpoint and endpoint categories for three perspectives - individualist (I), heirarchist (H) and egalitarian (E)

Midpoint-Abbreviation	Impact category unit	Endpoint impact category HH (DALY)	ED (species.yr)	RC ($)
CC	kg (CO_2 to air)	1.19 x 10^{-06} (I) 1.40 x 10^{-06} (H) 3.51 x 10^{-06} (E)	8.73x 10^{-06} (I+H) 18.8 x 10^{-06} (E)	0
OD	kg (CFC-11 to air)	See below	0	0
TA	kg (SO_2 to air)	0	1.52 x 10^{-09} (I) 5.8 x 10^{-09} (E) 14.2 x 10^{-09} (H)	
FE	kg (P to freshwater)	0	4.44 x 10^{-08}	0
ME	kg (N to freshwater)	0	0	0
HT	kg (1,4 DCB to urban air)	7.0 x 10^{-07} (I,H,E)	0	0
POF	kg (NMVOC to urban air)	3.9 x 10^{-08}	0	0
PMF	kg (PM_{10} to air)	2.6 x 10^{-04}	0	
TET	kg (1,4 DCB to ind, soil)	0		
FET	kg (1,4 DCB to freshwater)	0		
MET	kg (1,4 DCB to marine water)	0		
IR	kg (U235 to air)	1.64 x 10^{-08}		
ALO	m^2x.yr (agricultural land)	0		
ULO	m^2x.yr (urban land)	0		
NLT	m^2(natural land)	0		
WD	m^3 (water)	0		
MD	kg	0		
FD	kg	0		

Source: Goedkoop et al. (2013).

Chapter 5 - Supplementary material

Table B 5-1: Transport data used for transport sensitivity analysis - sunflowerseed oil system.

	Sea transport (tkm)	Road transport (tkm)
Base Case	2113.5	746.3
+15% Road	2113.5	858.2
+15% Sea	2430.5	746.3
+30% Sea	2747.6	746.3
+45%Sea	3064.6	746.3
+60% Sea	3381.6	746.3

tkm = the product of the number of tonnes transported and the distance travelled

Chapter 6 - Supplementary material

TableB6-1: Packaging calculations and assumptions for mayonnaise systems

Component		Assumption / calculation	Comment
Glass	Included	600 ml glass jar weighs 283g 1 tonne mayonnaise requires: 1000 kg/0.6 kg – 1667 jars Each weighing 0.283 kg = 1667 x 0.283 = 471.7kg clear glass	From weighing empty washed jar
Corrugated board	Included	Each tray carries 12 jars 1 tonne mayonnaise requires 1667 jars Therefore 139 trays Each weighs 29g =	From weighing cardboard tray
Packaging film	Included	Approximation of packing film required for wrap around tray and shrink wrap of pallet = 5 kg	Estimated from past experience of FMCG manufacture
Labels	Excluded		
Screw caps	Excluded	Composite material including steel, paint and rubberise inner	

Chapter 7 - Supplementary material

Table B 7-1: Data collected at Sutton Bonington trials to produce sunflowerseed oil bodies

Process input	Data collected during SB production trials	Lab data scaled to 1kg WOB production
Washed sunflowerseed	6.0 kg	4.615 kg
Water	24.0 kg	18.462 kg
$NaHCO_3$	0.605kg	0.465 kg
Roboqbo Power	3.092 kWh	2.378 kg
Centrifuge Power	6.23 kWh	4.792 kg
Wet Oil Bodies (WOB)	1.3kg	1.000 kg

Chapter 8 - Supplementary material

Table B 8-1: Basic LCI for OBM – Rape or sunflowerseed.

Process input	Quantity	Unit
Vinegar for mayonnaise	90	kg
Sodium chloride, powder, at plant/RER	5	kg
EUR-flat pallet/RER	1.75	piece
Packaging, corrugated board, mixed fibre, single wall, at plant/RER	4.028	kg
Packaging glass, white, at plant/RER	471.67	kg
Packaging film, LDPE, at plant/RER	5	kg
Tap water, at user/RER	16.2	kg
Natural gas, burned in Mini CHP plant/CH	3.5	GJ
WOB_SF-Whichever case	888.88	kg

Table B 8-2: Transport data used for investigation of OB processing at farm.

LCA input process	Rapeseed OBM system			Sunflowerseed OBM system		
	Transport distance	Transport input	Transport input - farm processing	Transport distance	Transport input	Transport input - farm processing
Seed transported →		4.615 t	1.000 t		4.615 t	1.000 t
Transport farm to OB extractor (road)	725 km	3346 tkm	725 tkm	338 km	1560 tkm	338 tkm
Transport farm to OB extractor (sea)				1756 km	8104 tkm	1756 tkm
Transport from extractor to food factory (sea)	390 km	390 tkm	390 tkm	390 km	390 tkm	390 tkm
Transport from extractor to food factory (road)	196 km	390 tkm	190 tkm	196 km	196 tkm	196 tkm

APPENDIX C. **SEED OIL LCA**

Chapter 5 - Supplementary Material

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Impact category	Unit	Rape seed conventional, at farm/DE U	Rape seed conventional, Barrois, at farm/FR U	Rape seed conventional, Saxony-Anhalt, at farm/DE U	Rape seed extensive, at farm/CH U	Rape seed IP, at farm/CH U	Rape Seed aggregate data set (no DE)
Climate change	kg CO_2 eq	1326.327	1273.627	775.7934	952.8784	915.8888	979.5468
Ozone depletion	kg CFC-11 eq	7.67E-05	7.44E-05	5.04E-05	0.000104	8.89E-05	7.95E-05
Human toxicity	kg 1,4-DB eq	303.8652	304.1684	134.4523	103.036	115.4268	164.2709
Photochemical oxidant formation	kg NMVOC	3.770862	3.398228	3.083612	3.252614	3.0059	3.185088
Particulate matter formation	kg PM10 eq	2.381017	2.852558	1.366078	3.522442	2.551671	2.573187
Ionising radiation	kg U235 eq	121.37	56.32928	50.98792	87.59106	89.02146	70.98243
Terrestrial acidification	kg SO_2 eq	10.49376	14.868	4.305535	21.11333	13.91576	13.55066
Freshwater eutrophication	kg P eq	0.626164	0.569915	0.306083	0.227053	0.320652	0.355926
Marine eutrophication	kg N eq	8.44185	22.03086	6.693336	17.18622	9.998302	13.97718
Terrestrial ecotoxicity	kg 1,4-DB eq	289.3193	103.6603	32.57971	1.029468	108.9049	61.54361
Freshwater ecotoxicity	kg 1,4-DB eq	63.65004	23.60774	11.20916	6.895061	23.02239	16.18359
Marine ecotoxicity	kg 1,4-DB eq	8.411553	4.753408	3.044943	2.449307	3.818349	3.516502
Agricultural land occupation	m^2a	2708.615	3609.182	2876.586	3389.902	2922.963	3199.658
Urban land occupation	m^2a	52.97258	6.06477	11.82691	49.34539	49.21324	29.11258
Natural land transformation	m^2	0.158874	0.115687	0.092831	0.114016	0.117159	0.109923
Water depletion	m^3	5.627368	4.087538	2.696073	1.401213	2.014623	2.549862
Metal depletion	kg Fe eq	63.73238	54.30903	52.47224	40.86225	40.13694	46.94512
Fossil depletion	kg oil eq	201.8657	158.2249	120.6789	123.7165	127.3037	132.481

Table C 5-1: Rapeseed cultivation datasets: comparison of characterised midpoint data.

	Characterised Data			Normalised Data		
Endpoint impact category	Unit	Rapeseed Oil	Sunflowerseed Oil	Unit	Rapeseed Oil	Sunflowerseed Oil
Climate change Human Health	DALY	3.18E-03	3.64E-03	None	1.58E-	1.80E-01
Ozone depletion	DALY	5.42E-07	4.45E-07	None	2.69E-	2.21E-05
Human toxicity	DALY	2.60E-04	1.66E-04	None	1.29E-	8.25E-03
Photochemical oxidant formation	DALY	3.33E-07	3.70E-07	None	1.65E-	1.84E-05
Particulate matter formation	DALY	1.38E-03	9.38E-04	None	6.85E-	4.65E-02
Ionising radiation	DALY	3.33E-06	4.03E-06	None	1.65E-	2.00E-04
Climate change Ecosystems	species.yr	1.80E-05	2.06E-05	None	1.03E-	1.18E-01
Terrestrial acidification	species.yr	1.55E-07	6.07E-08	None	8.90E-	3.47E-04
Freshwater eutrophication	species.yr	3.23E-08	3.35E-08	None	1.85E-	1.92E-04
Terrestrial ecotoxicity	species.yr	1.47E-05	1.38E-06	None	8.44E-	7.93E-03
Freshwater ecotoxicity	species.yr	8.21E-09	1.06E-08	None	4.70E-	6.05E-05
Marine ecotoxicity	species.yr	6.37E-12	6.00E-12	None	3.65E-	3.43E-08
Agricultural land occupation	species.yr	1.10E-04	2.79E-04	None	6.31E-	1.60E+00
Urban land occupation	species.yr	1.11E-06	1.87E-06	None	6.37E-	1.07E-02
Natural land transformation	species.yr	6.36E-07	5.97E-07	None	3.64E-	3.42E-03
Metal depletion	$	7.06E+00	7.07E+00	None	2.63E-	2.64E-04
Fossil depletion	$	6.71E+03	6.82E+03	None	2.50E-	2.54E-01

Table C 5-2: Analysis of Seed oil LCA at the endpoint level (ReCiPe(2008), European normalisation); economic allocation

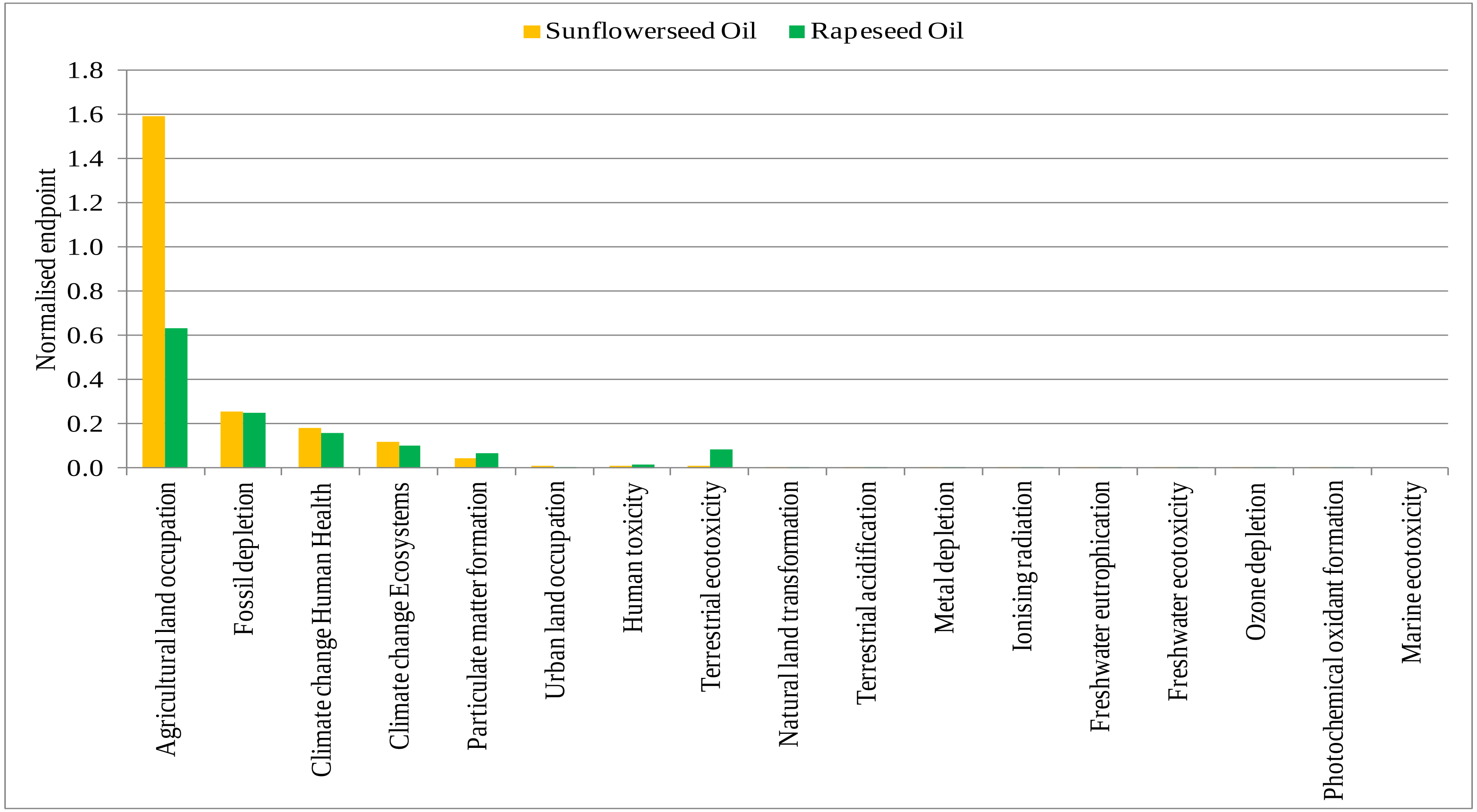

Figure C 5-1: Comparison of normalised endpoints for sunflower and rape seed oil systems, indicating most prominent impact categories.

	Rapeseed oil					Sunflowerseed oil				
	TOTAL	Cultivation	Extraction	Refining	Transport	TOTAL	Cultivation	Extraction	Refining	Transport
Agricultural land occupation	6.31E-01	6.31E-01	4.64E-05	6.32E-05	4.24E-05	1.60E+00	1.60E+00	4.77E-05	6.33E-05	3.64E-05
Fossil depletion	2.50E-01	1.49E-01	3.39E-02	3.01E-02	3.74E-02	1.18E-01	9.88E-02	6.95E-03	5.28E-03	6.84E-03
Climate change Human Health	1.58E-01	1.28E-01	1.03E-02	8.06E-03	1.18E-02	1.80E-01	1.51E-01	1.06E-02	8.07E-03	1.05E-02
Climate change Ecosystems	1.03E-01	8.34E-02	6.76E-03	5.27E-03	7.69E-03	2.54E-01	1.56E-01	3.48E-02	3.01E-02	3.29E-02
Terrestrial ecotoxicity	8.44E-02	8.44E-02	1.16E-06	8.59E-06	2.09E-05	6.05E-05	5.88E-05	3.64E-07	7.49E-07	5.80E-07
Particulate matter formation	6.85E-02	6.22E-02	1.23E-03	1.87E-03	3.18E-03	1.92E-04	1.75E-04	4.11E-06	8.61E-06	3.87E-06
Human toxicity	1.29E-02	1.07E-02	4.67E-04	1.02E-03	6.93E-04	8.25E-03	6.18E-03	4.10E-04	1.02E-03	6.32E-04
Urban land occupation	6.37E-03	6.04E-03	2.66E-05	8.00E-05	2.25E-04	2.00E-04	1.43E-04	2.13E-05	2.33E-05	1.23E-05
Natural land transformation	3.64E-03	2.69E-03	5.19E-05	3.62E-04	5.38E-04	3.43E-08	2.81E-08	1.22E-09	2.64E-09	2.38E-09
Terrestrial acidification	8.90E-04	8.44E-04	9.92E-06	1.72E-05	1.94E-05	2.64E-04	2.39E-04	1.44E-06	6.22E-06	1.69E-05
Metal depletion	2.63E-04	2.35E-04	1.40E-06	6.23E-06	2.08E-05	3.42E-03	2.54E-03	5.34E-05	3.39E-04	4.92E-04
Freshwater eutrophication	1.85E-04	1.68E-04	4.00E-06	8.97E-06	3.89E-06	2.21E-05	1.69E-05	3.18E-07	1.88E-06	2.98E-06
Ionising radiation	1.65E-04	1.08E-04	2.07E-05	2.31E-05	1.32E-05	4.65E-02	3.94E-02	1.27E-03	1.85E-03	4.05E-03
Freshwater ecotoxicity	4.70E-05	4.53E-05	3.54E-07	7.57E-07	6.36E-07	1.84E-05	1.55E-05	5.44E-07	4.56E-07	1.82E-06
Ozone depletion	2.69E-05	2.12E-05	3.09E-07	1.88E-06	3.47E-06	3.47E-04	2.92E-04	1.02E-05	1.71E-05	2.81E-05
Photochemical oxidant formation	1.65E-05	1.16E-05	2.90E-06	4.57E-07	1.60E-06	7.93E-03	7.90E-03	1.19E-06	8.48E-06	1.75E-05
Marine ecotoxicity	3.65E-08	3.02E-08	1.19E-09	2.67E-09	2.42E-09	1.07E-02	1.04E-02	2.74E-05	6.90E-05	1.84E-04
TOTAL	1.32E+00	1.16E+00	5.28E-02	4.69E-02	6.16E-02	2.23E+00	2.07E+00	5.42E-02	4.69E-02	5.57E-02
Contribution to total ➔		**87.8%**	**4.0%**	**3.6%**	**4.7%**		**93.0%**	**2.4%**	**2.1%**	**2.5%**

Table C 5-3: Normalised endpoint values with process contributions – European normalisation.

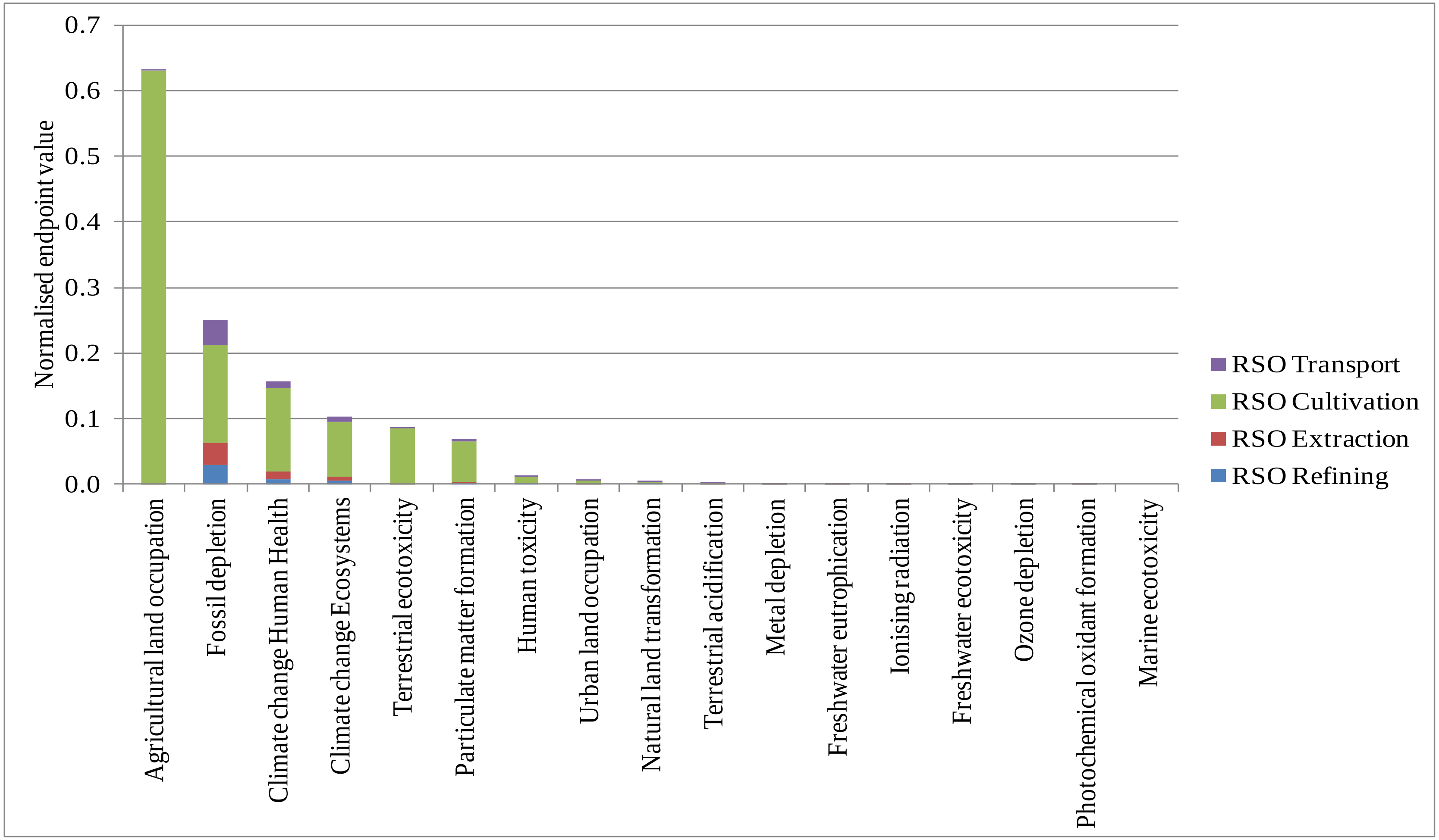

Figure C 5-2: Normalised endpoint analysis of rapeseed oil system, indicating most prominent impact categories.

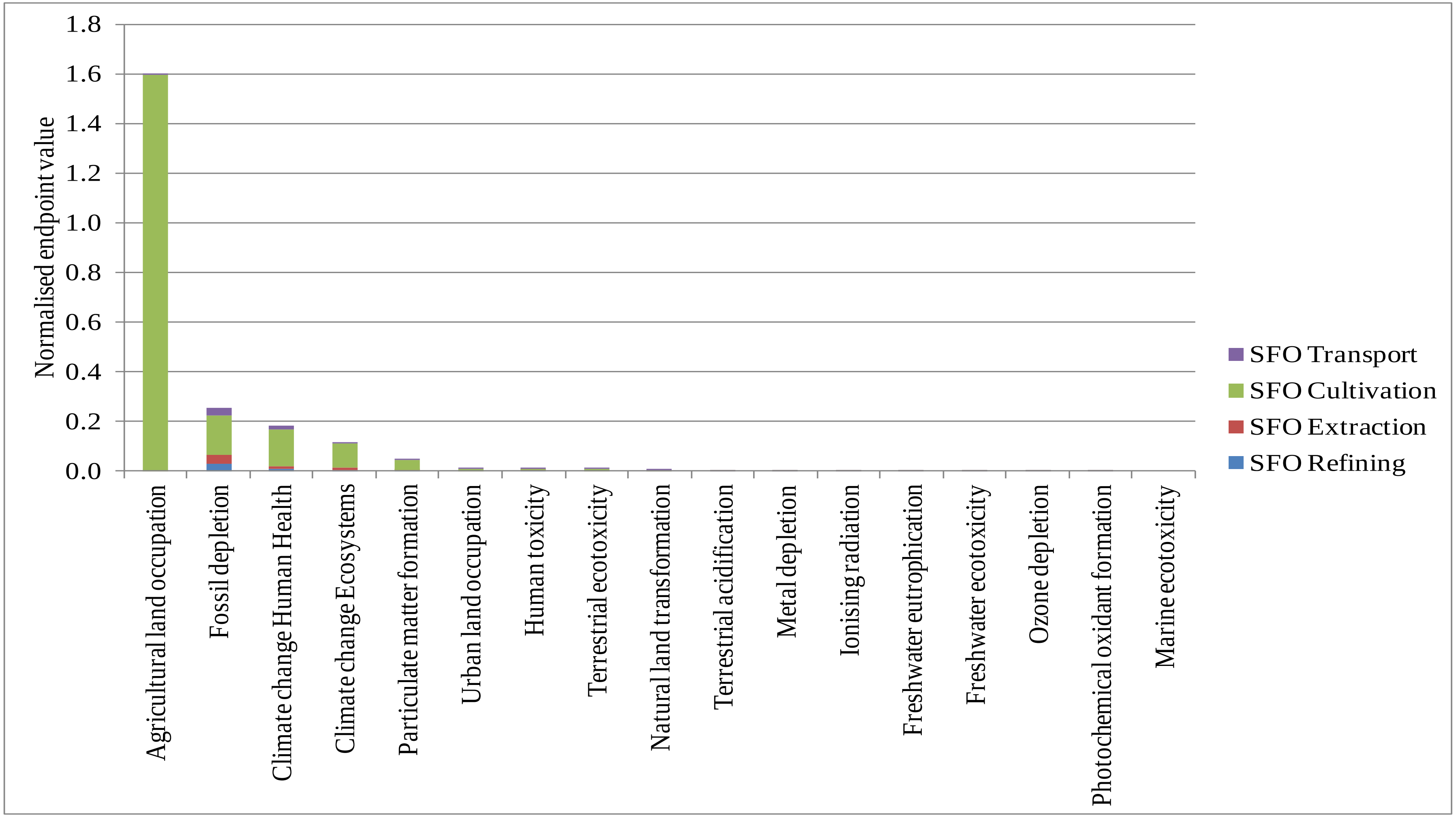

Figure C 5-3: Normalised endpoint analysis of sunflowerseed oil system, indicating most prominent impact categories.

	Rapeseed oil					Sunflowerseed oil				
	TOTAL	**Cultivation**	**Extraction**	**Refining**	**Transport**	**TOTAL**	**Cultivation**	**Extraction**	**Refining**	**Transport**
Agricultural land occupation	1.33E+00	1.33E+00	1.60E-04	2.17E-04	1.42E-04	3.35E+00	3.35E+00	1.64E-04	2.17E-04	1.22E-04
Climate change	2.03E-01	1.64E-01	1.33E-02	1.04E-02	1.51E-02	2.32E-01	1.94E-01	1.36E-02	1.04E-02	1.34E-02
Fossil depletion	2.51E-01	1.49E-01	3.40E-02	3.02E-02	3.75E-02	2.55E-01	1.57E-01	3.49E-02	3.02E-02	3.30E-02
Freshwater ecotoxicity	2.90E+00	2.79E+00	2.20E-02	4.67E-02	3.93E-02	3.73E+00	3.63E+00	2.26E-02	4.62E-02	3.58E-02
Freshwater eutrophication	1.77E+00	1.61E+00	3.83E-02	8.59E-02	3.73E-02	1.83E+00	1.67E+00	3.94E-02	8.25E-02	3.71E-02
Human toxicity	6.26E-01	5.20E-01	2.27E-02	4.95E-02	3.37E-02	4.01E-01	3.01E-01	1.99E-02	4.98E-02	3.07E-02
Ionising radiation	3.25E-02	2.13E-02	4.07E-03	4.54E-03	2.59E-03	3.94E-02	2.82E-02	4.18E-03	4.58E-03	2.42E-03
Marine ecotoxicity	9.38E-01	7.76E-01	3.08E-02	6.85E-02	6.22E-02	8.82E-01	7.22E-01	3.17E-02	6.78E-02	6.11E-02
Marine eutrophication	2.60E+00	2.59E+00	1.41E-03	1.48E-03	2.80E-03	8.46E+00	8.45E+00	1.44E-03	1.49E-03	3.27E-03
Metal depletion	1.38E-01	1.23E-01	7.36E-04	3.27E-03	1.09E-02	1.39E-01	1.26E-01	7.57E-04	3.27E-03	8.90E-03
Natural land transformation	2.02E+00	1.28E+00	9.71E-02	2.60E-01	3.82E-01	2.21E+00	1.50E+00	9.99E-02	2.60E-01	3.53E-01
Ozone depletion	8.75E-03	6.77E-03	1.15E-04	6.56E-04	1.21E-03	7.56E-03	5.75E-03	1.19E-04	6.55E-04	1.04E-03
Particulate matter formation	3.56E-01	3.24E-01	6.40E-03	9.72E-03	1.65E-02	2.42E-01	2.05E-01	6.58E-03	9.65E-03	2.10E-02
Photochemical oxidant formation	1.61E-01	1.12E-01	2.81E-02	4.45E-03	1.56E-02	1.79E-01	1.51E-01	5.29E-03	4.44E-03	1.77E-02
Terrestrial acidification	7.80E-01	7.39E-01	8.68E-03	1.50E-02	1.69E-02	3.04E-01	2.56E-01	8.93E-03	1.49E-02	2.46E-02
Terrestrial ecotoxicity	1.41E+01	1.41E+01	1.95E-04	1.44E-03	3.50E-03	1.33E+00	1.32E+00	2.00E-04	1.42E-03	2.93E-03
Urban land occupation	1.42E-01	1.34E-01	5.92E-04	1.78E-03	4.99E-03	2.39E-01	2.32E-01	6.08E-04	1.53E-03	4.10E-03
Water depletion	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
TOTAL	2.83E+01	2.67E+01	3.09E-01	5.94E-01	6.82E-01	2.38E+01	2.23E+01	2.90E-01	5.89E-01	6.50E-01
Contribution to total		**94.4%**	**1.1%**	**2.1%**	**2.4%**		**93.6%**	**1.2%**	**2.5%**	**2.7%**

Table C 5-4: Normalised midpoint values with process contributions – European normalisation.

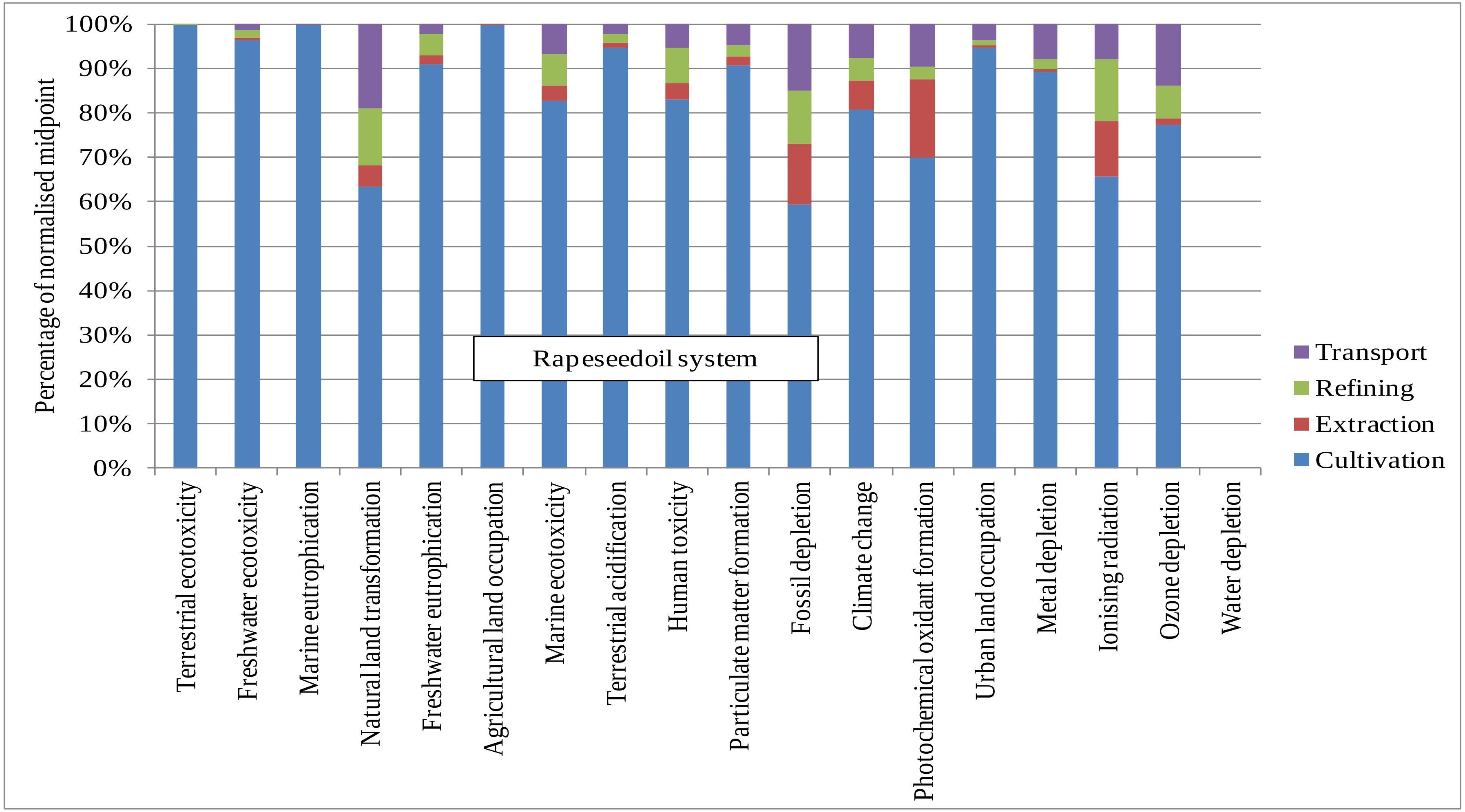

Figure C 5-4: Percentage normalised midpoints – rapeseed oil system.

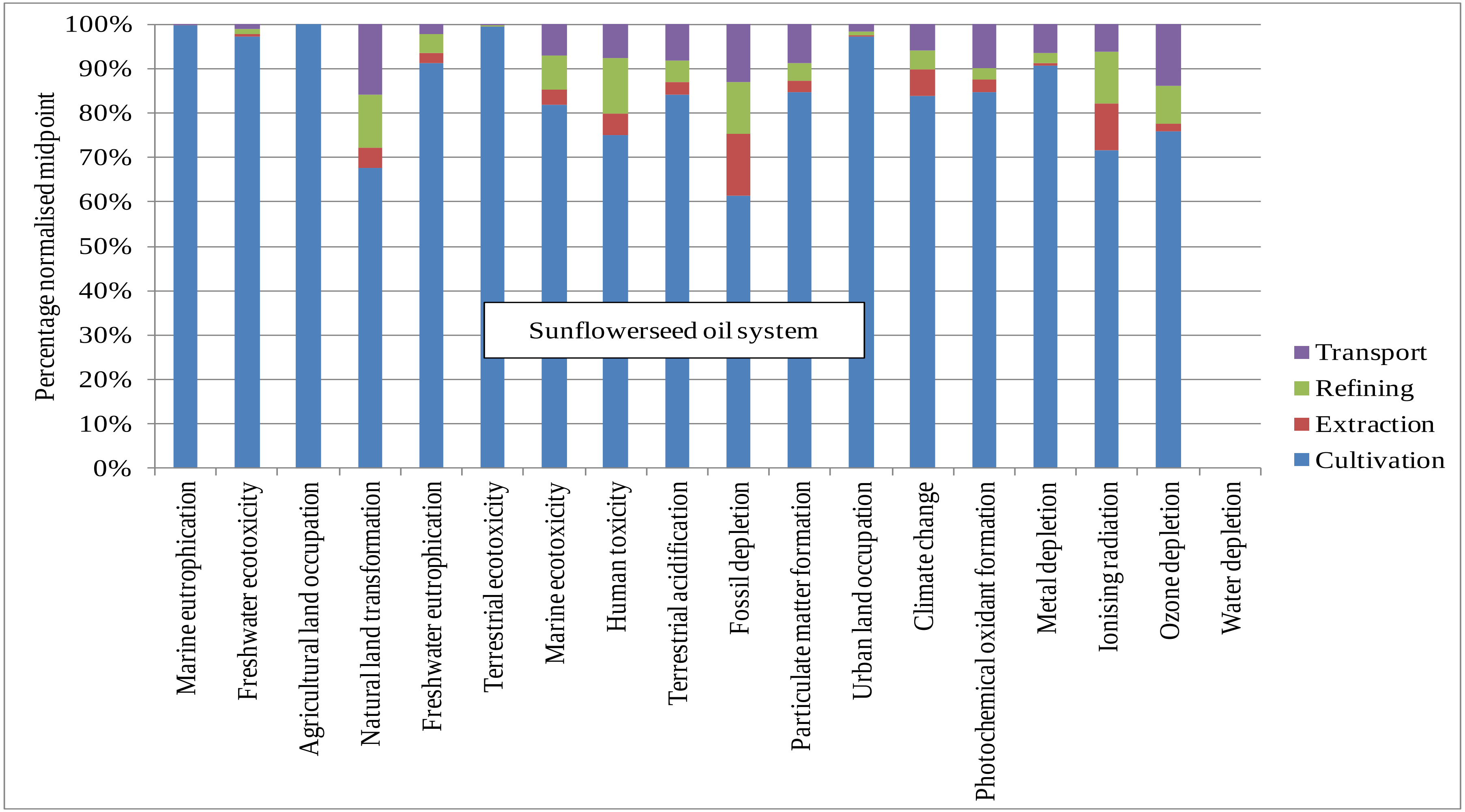

Figure C 5-5: Percentage normalised midpoints - sunflowerseed oil system.

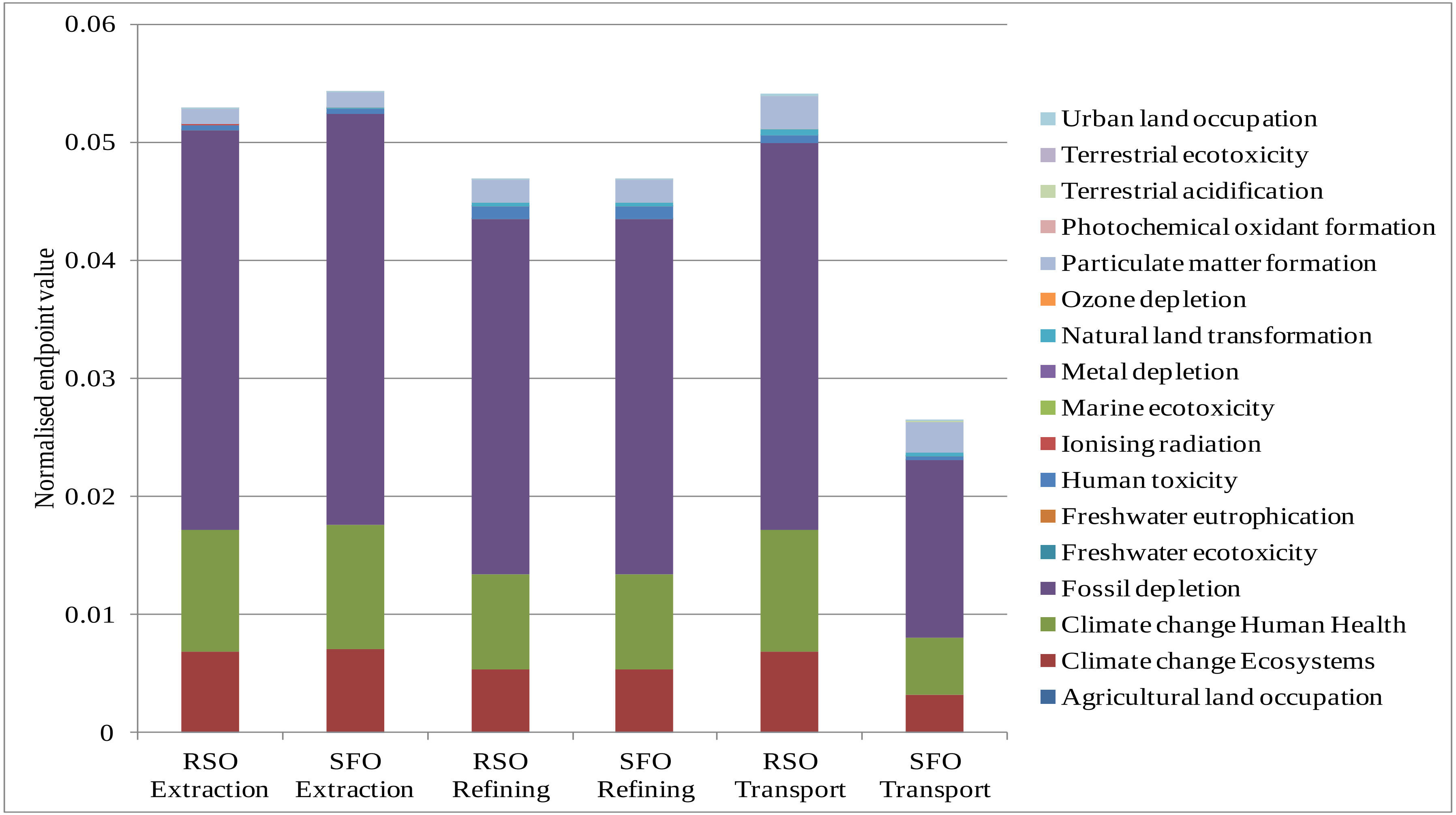

Figure C 5-6: Process breakdown for normalised endpoint values – gate-to-gate (G2G) analysis.

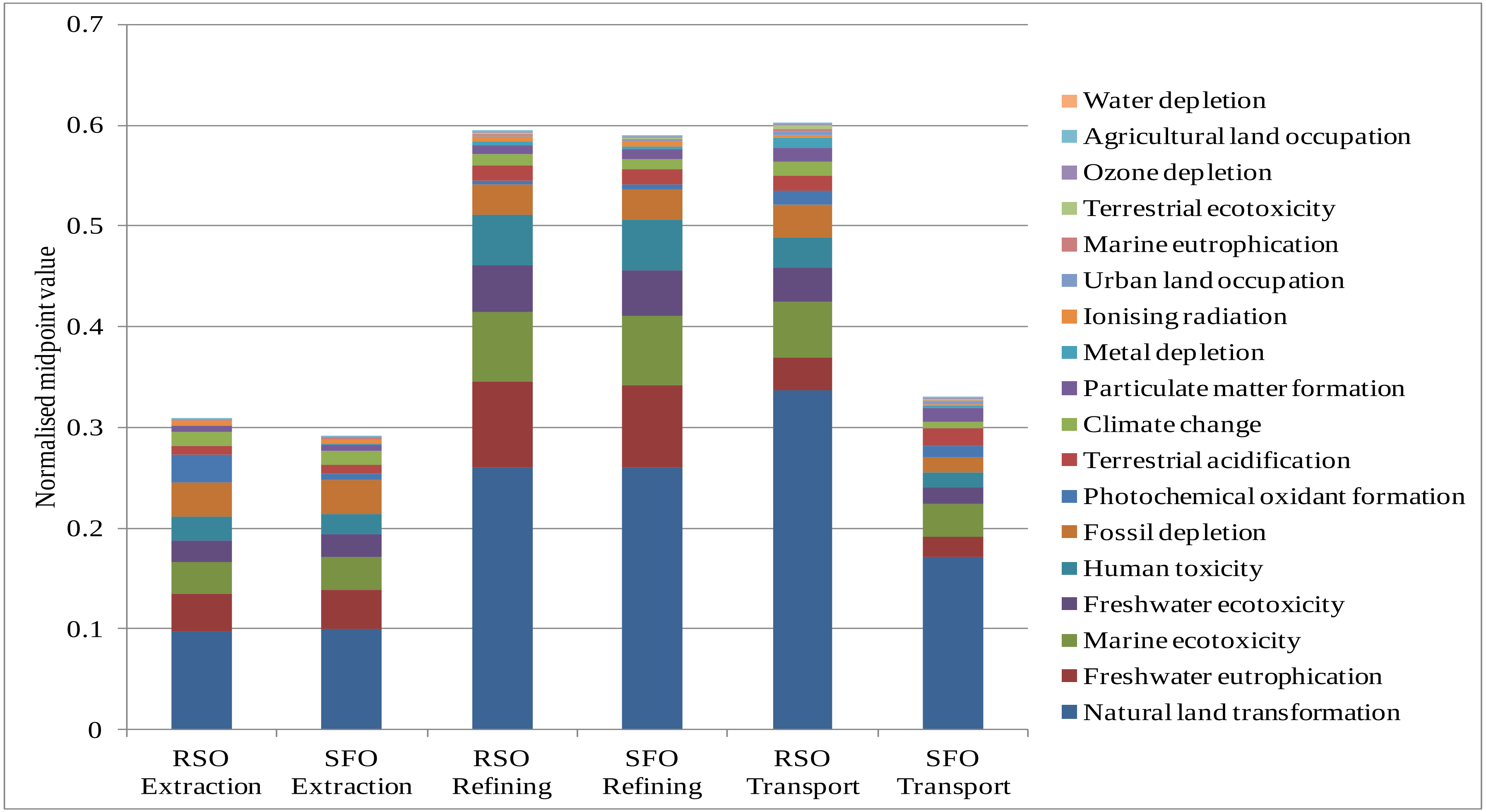

Figure C 5-7: Process breakdown for normalised midpoint values – gate-to-gate (G2G) analysis.

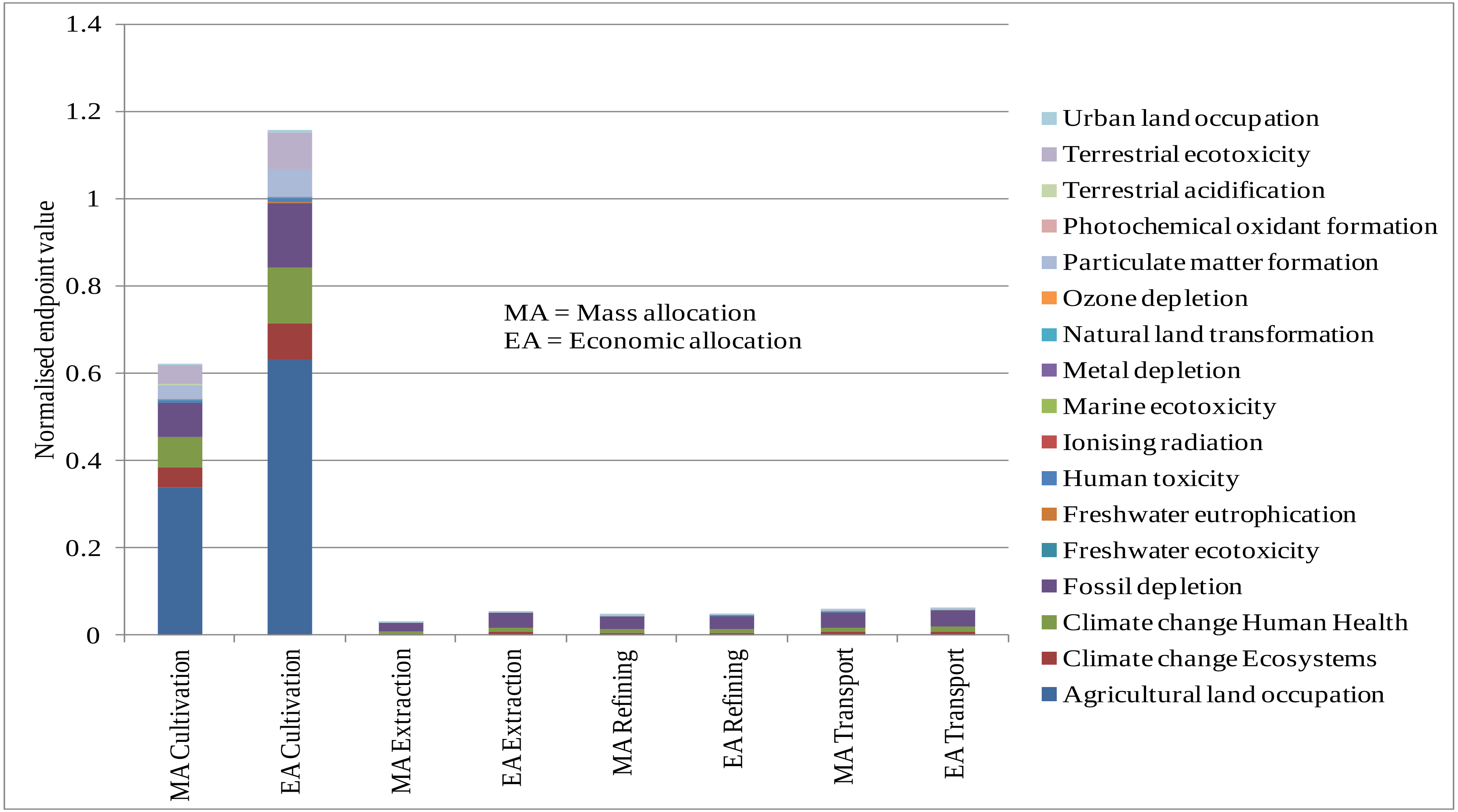

Figure C 5-8: Effect of changing allocation methodology on rapeseed oil system.

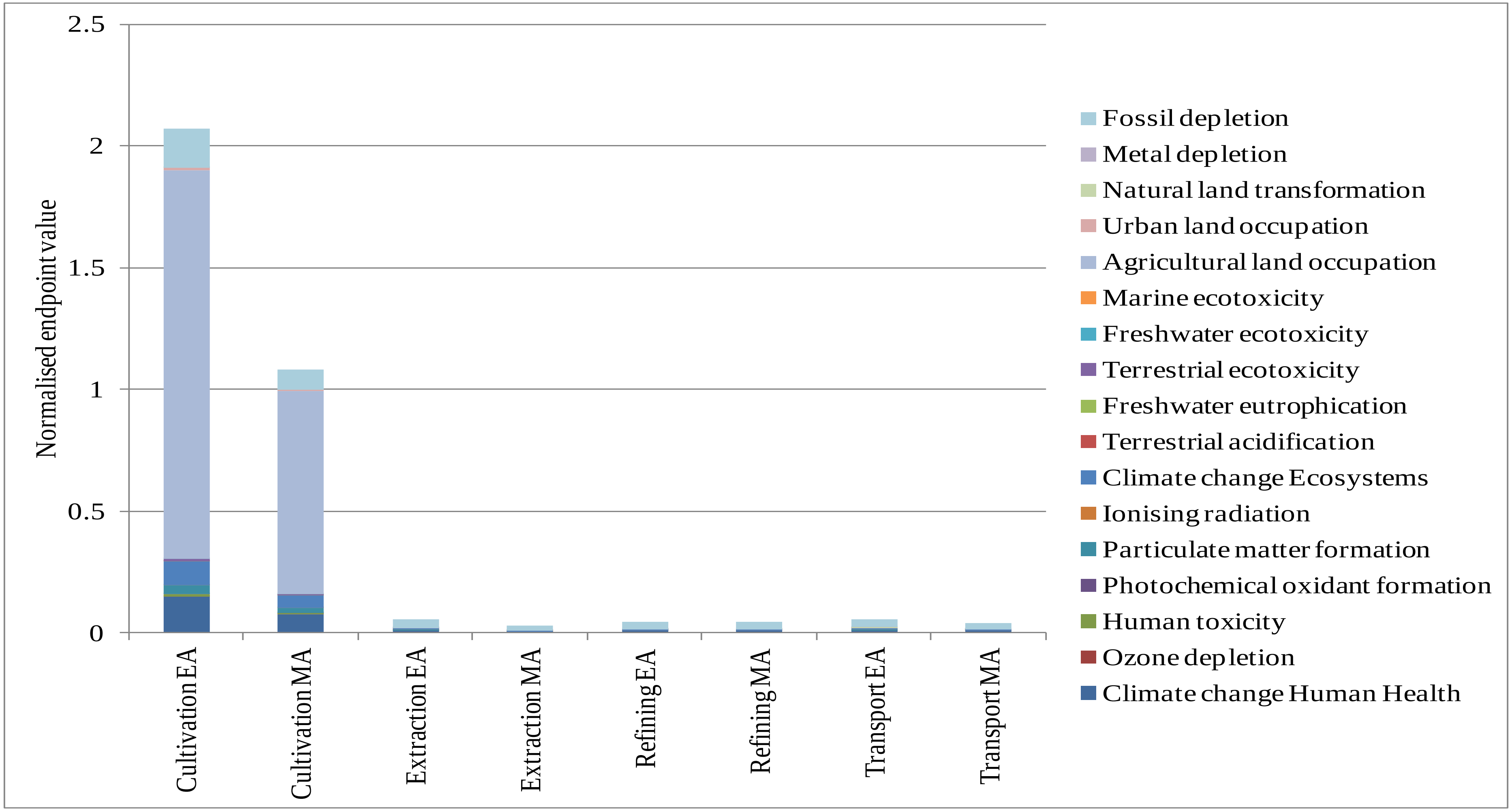

Figure C 5-9: Effect of changing allocation methodology on sunflowerseed oil system.

		Rapeseed oil EA	Rapeseed oil MA	Percentage change	Sunflowerseed oil EA	Sunflowerseed oil MA	Percentage change
Climate change	kg CO_2 eq	2271	1340	-41.01%	2597	1445	-44.37%
Ozone depletion	kg CFC-11 eq	0.0002	0.0001	-37.46%	0.0002	0.0001	-40.87%
Human toxicity	kg 1,4-DB eq	371	220	-40.58%	238	143	-39.81%
Photochemical oxidant formation	kg NMVOC	8.5	5.0	-41.13%	9.5	6.0	-36.84%
Particulate matter formation	kg PM10 eq	5.3	3.0	-43.15%	3.6	2.0	-43.20%
Ionising radiation	kg U235 eq	203	128	-36.86%	246	145	-40.92%
Terrestrial acidification	kg SO_2 eq	26.8	14.9	-44.48%	10.5	6.0	-42.75%
Freshwater eutrophication	kg P eq	0.7	0.4	-43.23%	0.8	0.4	-45.11%
Marine eutrophication	kg N eq	26.3	14.1	-46.14%	85.5	44.8	-47.67%
Terrestrial ecotoxicity	kg 1,4-DB eq	115	62.1	-46.20%	10.9	5.7	-47.59%
Freshwater ecotoxicity	kg 1,4-DB eq	31.5	17.4	-44.96%	40.6	21.6	-46.90%
Marine ecotoxicity	kg 1,4-DB eq	8.0	4.8	-40.35%	7.5	4.3	-42.46%
Agricultural land occupation	m^2a	6003	3230	-46.20%	15158	7931	-47.68%
Urban land occupation	m^2a	57.6	32.1	-44.27%	97.1	51.4	-47.06%
Natural land transformation	m^2	0.3	0.2	-33.01%	0.4	0.2	-38.65%
Water depletion	m^3	7.4	4.9	-33.06%	6.2	4.1	-33.95%
Metal depletion	kg Fe eq	98.7	57.2	-42.06%	98.9	54.0	-45.39%
Fossil depletion	kg oil eq	418	271	-35.00%	424	257	-39.48%

Table C 5-5: Percentage change in characterised midpoints through moving from economic to mass allocation.

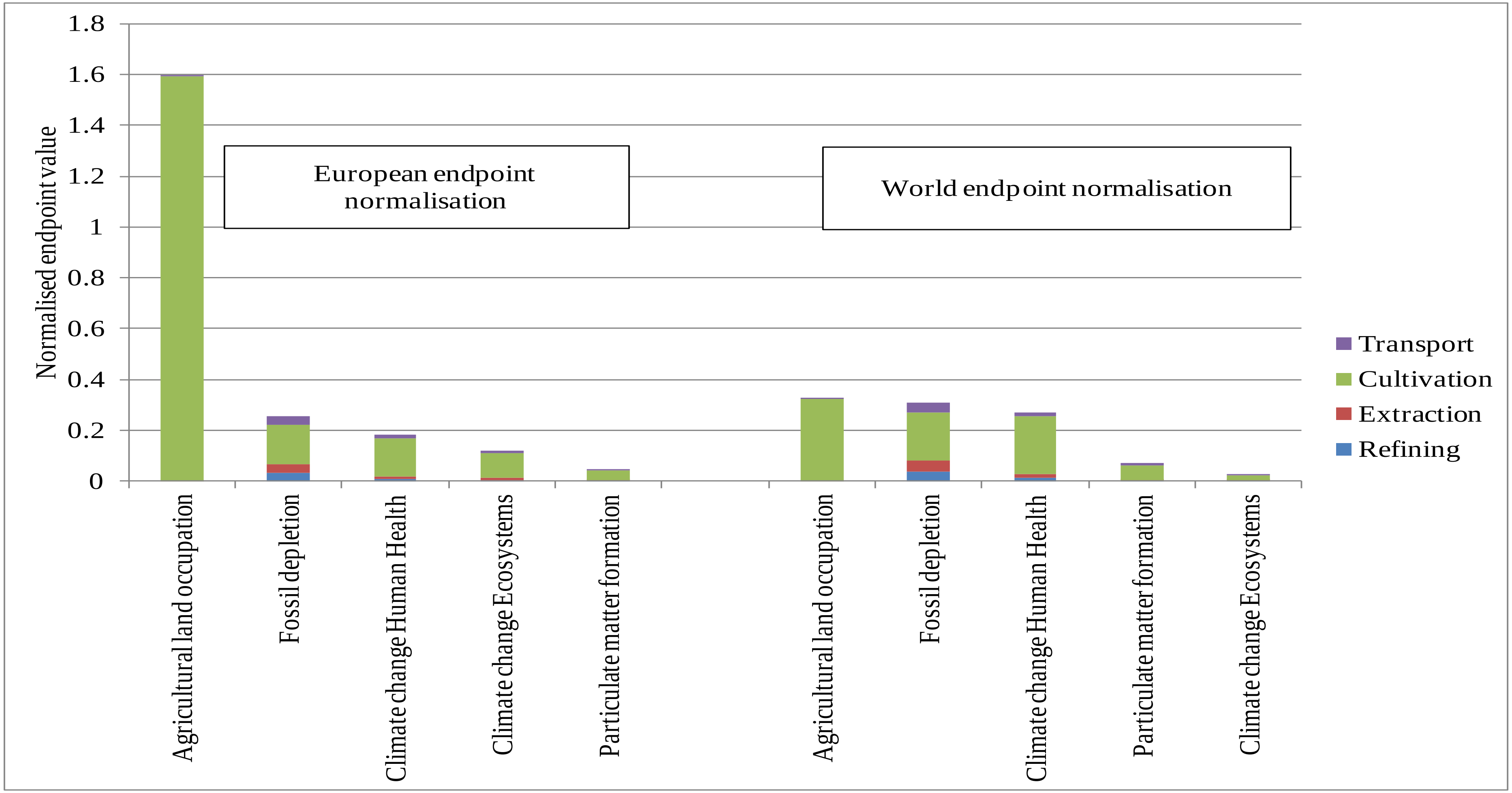

Figure C 5-10: Top five normalised endpoint categories for sunflowerseed oil system with different normalisation values. .

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APPENDIX D. **MAYONNAISE LCA**

Chapter 6 - Supplementary Material

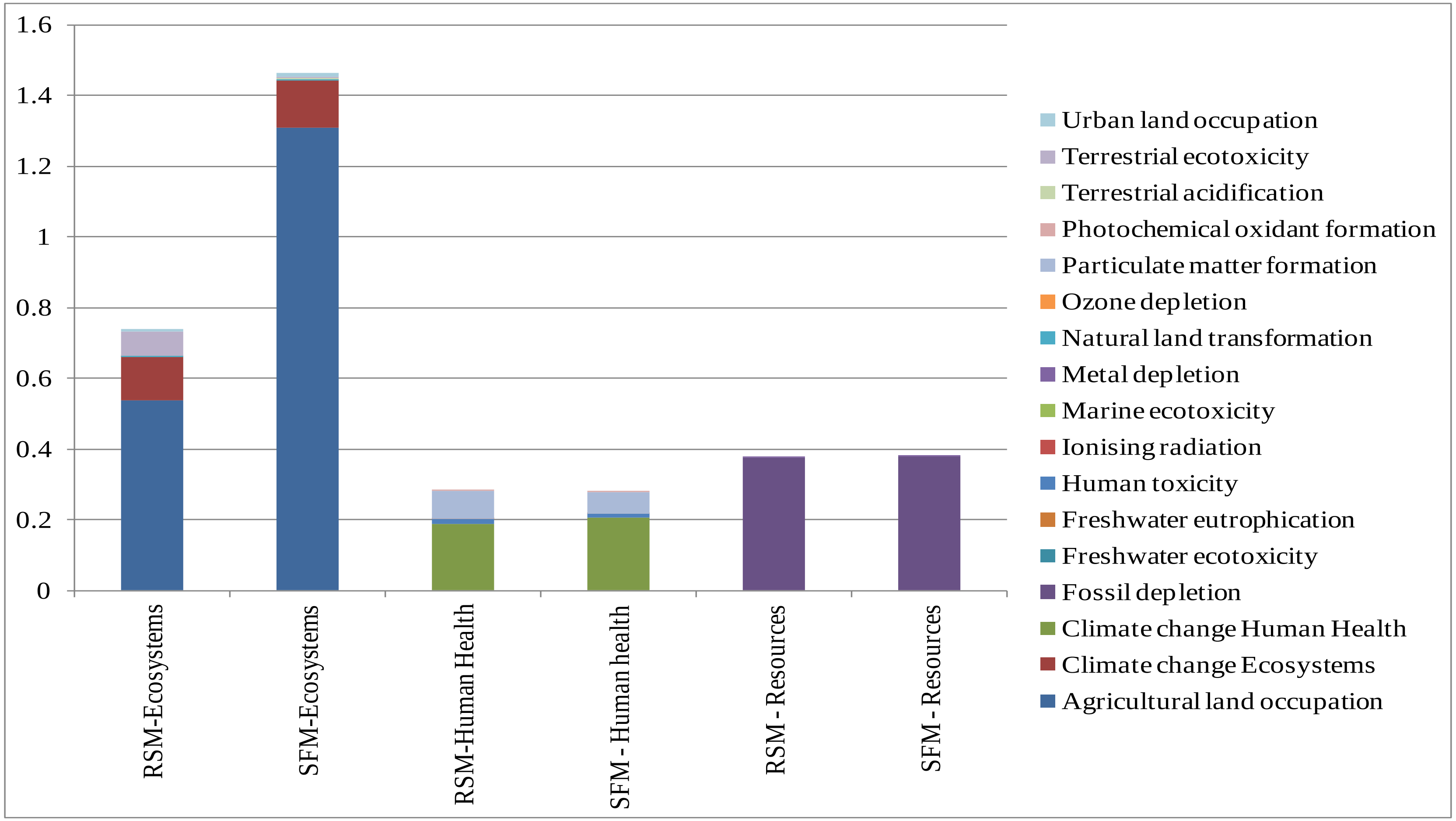

Figure D 6-1: Normalised endpoint results for mayonnaise systems. RSM - rapeseed mayonnaise; SFM - sunflowerseed mayonnaise.

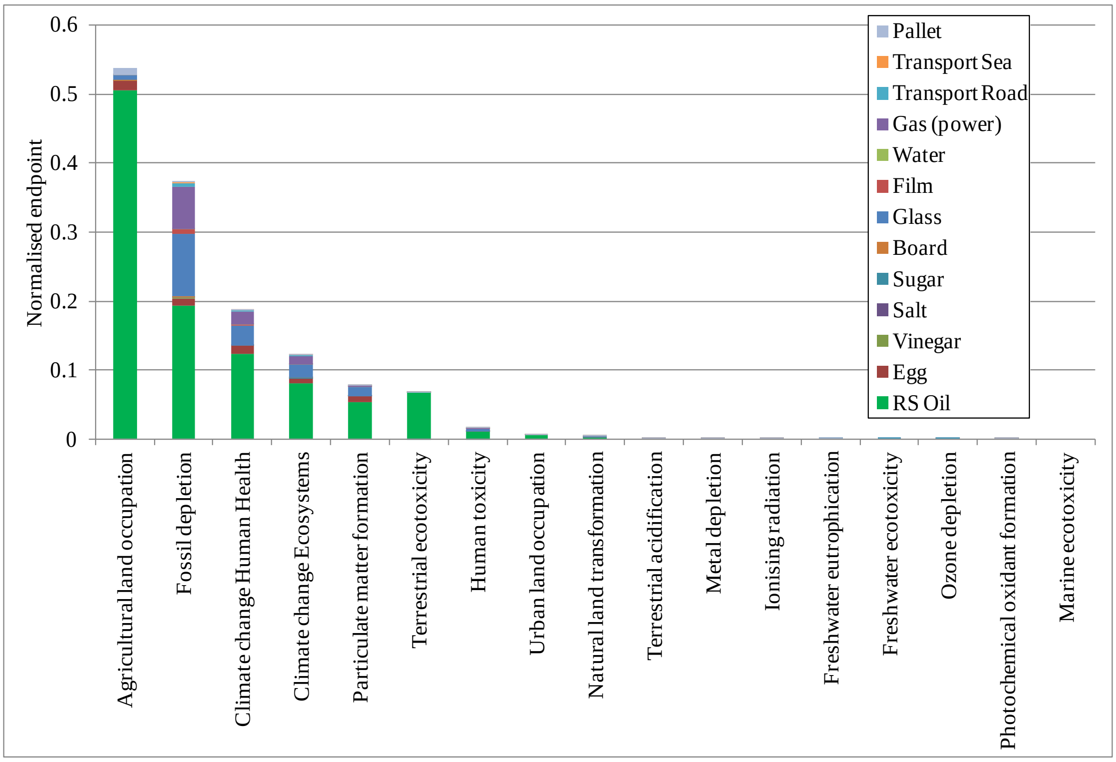

Figure D 6-2: Normalised endpoint results for rapeseed mayonnaise system.

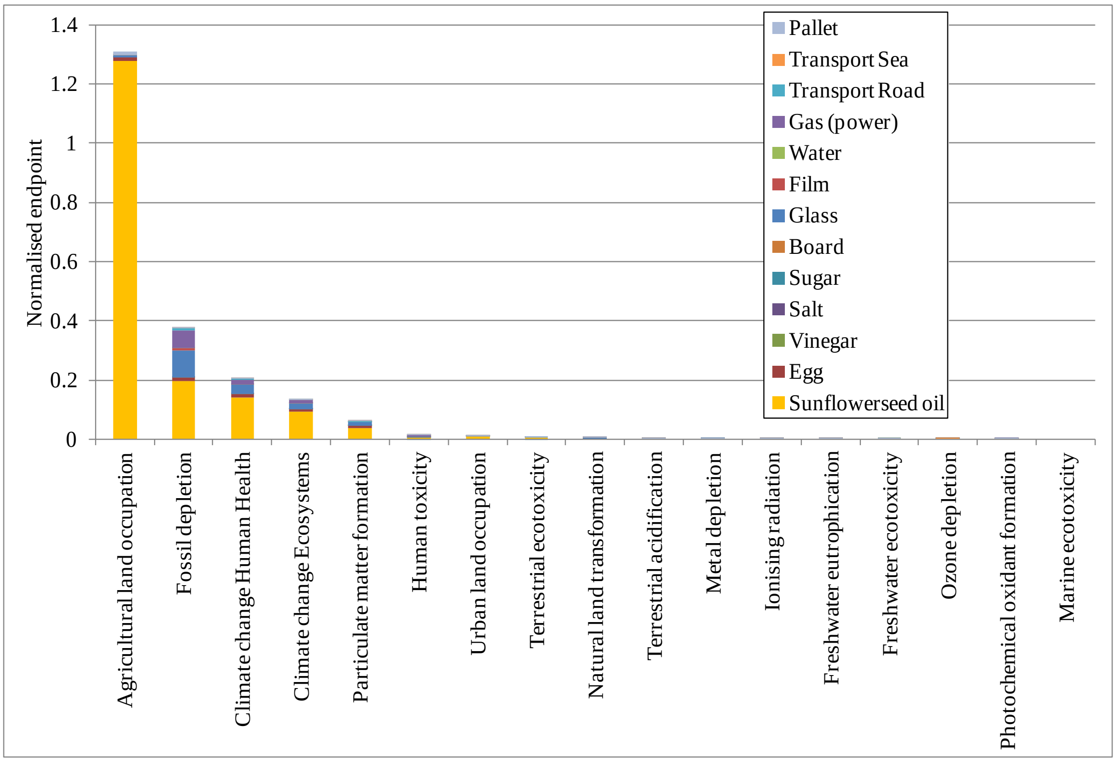

Figure D 6-3: Normalised endpoint results for sunflowerseed mayonnaise system.

Impact category	Unit	Total	Rapeseed oil	Egg	Vinegar	Salt	Sugar	Packaging board	Packaging glass	Packaging film	Water	Power (Mini CHP)	Transport	Pallet
Agricultural land occupation	m^2a	5209.7	4802.5	126.5	0.05	0.13	9.31	9.53	118.4	0.83	0.14	0.18	0.11	142.1
Fossil depletion	kg oil eq	624.2	323.2	16.2	1.99	0.50	1.13	1.51	152.2	9.38	0.53	101.7	10.8	5.17
Climate change	kg CO_2 eq	2693.7	1787.5	157.5	2.80	1.80	5.06	4.59	418.8	13.5	2.00	260.1	29.5	10.7
Particulate matter formation	kg PM_{10} eq	6.04	4.19	0.63	0.00	0.00	0.01	0.01	1.02	0.02	0.00	0.07	0.06	0.02
Terrestrial ecotoxicity	kg 1,4-DB eq	92.5	92.4	0.00	0.00	0.00	0.00	0.00	0.12	0.00	0.00	0.00	0.00	0.00
Human toxicity	kg 1,4-DB eq	451.7	293.1	1.45	1.25	1.91	-0.98	1.43	136.8	1.46	1.45	6.05	3.54	4.19
Urban land occupation	m^2a	51.9	45.7	0.00	0.01	0.02	0.04	0.17	3.63	0.03	0.09	0.16	0.33	1.62
Natural land transformation	m^2a	0.48	0.25	0.00	0.00	0.00	0.00	0.00	0.14	0.00	0.00	0.06	0.01	0.01
Terrestrial acidification	kg SO_2 eq	29.5	21.3	4.41	0.01	0.01	0.05	0.02	3.26	0.05	0.01	0.22	0.15	0.05
Metal depletion	kg Fe eq	96.7	77.7	0.32	0.19	0.56	0.24	0.23	11.9	0.11	0.12	2.42	1.28	1.67
Ionising radiation	kg U_{235} eq	244.7	159.6	1.07	1.07	0.94	1.77	0.85	69.0	1.49	1.24	1.77	2.91	3.00
Freshwater eutrophication	kg P eq	0.72	0.58	0.01	0.00	0.00	0.00	0.00	0.10	0.00	0.00	0.00	0.00	0.00
Freshwater ecotoxicity	kg 1,4-DB eq	27.6	25.1	0.06	0.03	0.03	0.04	0.04	1.96	0.04	0.04	0.10	0.08	0.09
Ozone depletion	kg CFC-11 eq	0.0003	0.0001	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Photochemical oxidant formation	kg NMVOC	9.59	6.65	0.34	0.01	0.00	0.02	0.02	1.91	0.05	0.01	0.33	0.17	0.08
Marine ecotoxicity	kg 1,4-DB eq	8.88	6.27	0.03	0.03	0.04	0.01	0.03	2.06	0.03	0.03	0.17	0.10	0.09
Marine eutrophication	kg N eq	23.1	21.0	1.95	0.00	0.00	0.04	0.00	0.10	0.00	0.00	0.01	0.01	0.00
Water depletion	m^3	17.5	5.79	0.14	0.05	0.06	0.09	0.09	3.82	0.03	7.13	0.08	0.11	0.10

Table D6-1: Breakdown of midpoint impacts by processing stage. Rapeseed oil mayonnaise.

		Total	SF Oil	Egg	Vinegar	Salt	Sugar	Board	Glass	Film	Water	Gas (power)	Transport	Pallet
Climate change	kg CO_2 eq	2955.24	2048.95	157.52	2.80	1.80	5.06	4.59	418.77	13.48	2.00	260.14	29.46	10.66
Ozone depletion	kg CFC-11 eq	0.0002	0.0001	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.00	0.0000
Human toxicity	kg 1,4-DB eq	346.84	188.30	1.45	1.25	1.91	-0.98	1.43	136.80	1.46	1.45	6.05	3.54	4.19
Photochemical oxidant formation	kg NMVOC	11.37	8.43	0.34	0.01	0.00	0.02	0.02	1.91	0.05	0.01	0.33	0.17	0.08
Particulate matter formation	kg PM10 eq	4.67	2.83	0.63	0.00	0.00	0.01	0.01	1.02	0.02	0.00	0.07	0.06	0.02
Ionising radiation	kg U235 eq	279.01	193.91	1.07	1.07	0.94	1.77	0.85	68.97	1.49	1.24	1.77	2.91	3.00
Terrestrial acidification	kg SO2 eq	16.45	8.22	4.41	0.01	0.01	0.05	0.02	3.26	0.05	0.01	0.22	0.15	0.05
Freshwater eutrophication	kg P eq	0.74	0.60	0.01	0.00	0.00	0.00	0.00	0.10	0.00	0.00	0.00	0.00	0.00
Marine eutrophication	kg N eq	70.58	68.46	1.95	0.00	0.00	0.04	0.00	0.10	0.00	0.00	0.01	0.01	0.00
Terrestrial ecotoxicity	kg 1,4-DB eq	8.86	8.72	0.00	0.00	0.00	0.00	0.00	0.12	0.00	0.00	0.00	0.00	0.00
Freshwater ecotoxicity	kg 1,4-DB eq	34.91	32.42	0.06	0.03	0.03	0.04	0.04	1.96	0.04	0.04	0.10	0.08	0.09
Marine ecotoxicity	kg 1,4-DB eq	8.51	5.90	0.03	0.03	0.04	0.01	0.03	2.06	0.03	0.03	0.17	0.10	0.09
Agricultural land occupation	m^2a	12538.99	12131.74	126.48	0.05	0.13	9.31	9.53	118.39	0.83	0.14	0.18	0.11	142.09
Urban land occupation	m^2a	83.45	77.34	0.00	0.01	0.02	0.04	0.17	3.63	0.03	0.09	0.16	0.33	1.62
Natural land transformation	m^2	0.50	0.27	0.00	0.00	0.00	0.00	0.00	0.14	0.00	0.00	0.06	0.01	0.01
Water depletion	m3	16.53	4.84	0.14	0.05	0.06	0.09	0.09	3.82	0.03	7.13	0.08	0.11	0.10
Metal depletion	kg Fe eq	96.84	77.85	0.32	0.19	0.56	0.24	0.23	11.86	0.11	0.12	2.42	1.28	1.67
Fossil depletion	kg oil eq	629.74	328.72	16.16	1.99	0.50	1.13	1.51	152.18	9.38	0.53	101.69	10.78	5.17

Table D6-2: Process contributions to characterised midpoints - sunflowerseed oil mayonnaise.

	Rapeseed oil	Egg	Vinegar	Salt	Sugar	Packaging board	Packaging glass	Packaging film	Water	Power (Mini CHP)	Transport	Pallet
	Percentage contribution to characterised midpoint value.											
Climate change	66.4	5.8	0.1	0.1	0.2	0.2	15.5	0.5	0.1	9.7	1.1	0.4
Ozone depletion	55.4	10.0	0.2	0.0	0.2	0.2	17.4	0.1	0.0	14.7	1.7	0.3
Human toxicity	64.8	0.3	0.3	0.4	-0.2	0.3	30.2	0.3	0.3	1.3	0.8	0.9
Photochemical oxidant formation	69.3	3.5	0.1	0.0	0.2	0.2	20.0	0.6	0.1	3.4	1.8	0.9
Particulate matter formation	69.5	10.5	0.1	0.0	0.2	0.1	16.9	0.3	0.1	1.2	0.9	0.4
Ionising radiation	65.2	0.4	0.4	0.4	0.7	0.3	28.2	0.6	0.5	0.7	1.2	1.2
Terrestrial acidification	72.2	14.9	0.0	0.0	0.2	0.1	11.1	0.2	0.0	0.8	0.5	0.2
Freshwater eutrophication	80.9	1.9	0.2	0.3	0.2	0.2	14.3	0.3	0.2	0.6	0.4	0.6
Marine eutrophication	90.8	8.4	0.0	0.0	0.2	0.0	0.5	0.0	0.0	0.0	0.0	0.0
Terrestrial ecotoxicity	99.8	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0
Freshwater ecotoxicity	91.0	0.2	0.1	0.1	0.1	0.1	7.1	0.1	0.1	0.3	0.3	0.3
Marine ecotoxicity	70.6	0.3	0.3	0.4	0.2	0.3	23.2	0.4	0.3	1.9	1.1	1.0
Agricultural land occupation	92.2	2.4	0.0	0.0	0.2	0.2	2.3	0.0	0.0	0.0	0.0	2.7
Urban land occupation	88.2	0.0	0.0	0.0	0.1	0.3	7.0	0.0	0.2	0.3	0.6	3.1
Natural land transformation	52.1	0.0	0.3	0.1	0.1	0.4	29.4	0.1	0.1	11.9	2.3	3.1
Water depletion	33.1	0.8	0.3	0.3	0.5	0.5	21.8	0.2	40.8	0.4	0.6	0.6
Metal depletion	80.4	0.3	0.2	0.6	0.2	0.2	12.3	0.1	0.1	2.5	1.3	1.7
Fossil depletion	51.8	2.6	0.3	0.1	0.2	0.2	24.4	1.5	0.1	16.3	1.7	0.8

Table D6-3: Percentage contribution of process stages to characterised midpoint value - Rapeseed oil mayonnaise

	Sunflowerseed oil	Egg	Vinegar	Salt	Sugar	Packaging board	Packaging glass	Packaging film	Water	Power (Mini CHP)	Transport	Pallet
	Percentage contribution to characterised midpoint value.											
Climate change	69.33	5.33	0.09	0.06	0.17	0.16	14.17	0.46	0.07	8.80	1.00	0.36
Ozone depletion	51.64	10.80	0.21	0.05	0.17	0.19	18.82	0.07	0.04	15.91	1.81	0.30
Human toxicity	54.29	0.42	0.36	0.55	-0.28	0.41	39.44	0.42	0.42	1.74	1.02	1.21
Photochemical oxidant formation	74.14	2.99	0.09	0.04	0.14	0.14	16.83	0.47	0.05	2.87	1.51	0.73
Particulate matter formation	60.59	13.51	0.07	0.06	0.22	0.13	21.83	0.33	0.07	1.54	1.20	0.46
Ionising radiation	69.50	0.38	0.38	0.34	0.63	0.30	24.72	0.54	0.45	0.63	1.04	1.08
Terrestrial acidification	50.00	26.80	0.06	0.05	0.29	0.09	19.85	0.28	0.05	1.36	0.89	0.28
Freshwater eutrophication	81.42	1.82	0.20	0.24	0.16	0.23	13.93	0.28	0.21	0.55	0.39	0.57
Marine eutrophication	97.00	2.76	0.00	0.00	0.05	0.01	0.15	0.00	0.00	0.01	0.01	0.00
Terrestrial ecotoxicity	98.41	0.03	0.01	0.00	-0.02	0.02	1.39	0.00	0.00	0.05	0.05	0.05
Freshwater ecotoxicity	92.87	0.17	0.07	0.10	0.10	0.10	5.63	0.10	0.11	0.28	0.22	0.24
Marine ecotoxicity	69.34	0.32	0.30	0.42	0.17	0.34	24.24	0.40	0.34	1.96	1.16	1.00
Agricultural land occupation	96.75	1.01	0.00	0.00	0.07	0.08	0.94	0.01	0.00	0.00	0.00	1.13
Urban land occupation	92.67	0.00	0.02	0.03	0.05	0.21	4.35	0.03	0.11	0.19	0.40	1.94
Natural land transformation	54.44	0.00	0.32	0.07	0.14	0.42	27.91	0.09	0.13	11.31	2.19	2.98
Water depletion	29.25	0.86	0.31	0.35	0.56	0.53	23.09	0.20	43.13	0.47	0.64	0.62
Metal depletion	80.39	0.33	0.19	0.58	0.25	0.23	12.25	0.11	0.12	2.50	1.32	1.73
Fossil depletion	52.20	2.57	0.32	0.08	0.18	0.24	24.17	1.49	0.08	16.15	1.71	0.82

Table D6-4: Percentage contribution of process stages to characterised midpoint value - Sunflowerseed oil mayonnaise

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APPENDIX E. **OIL-BODY PRODUCTION LCA**

Chapter 7 - Supplementary material

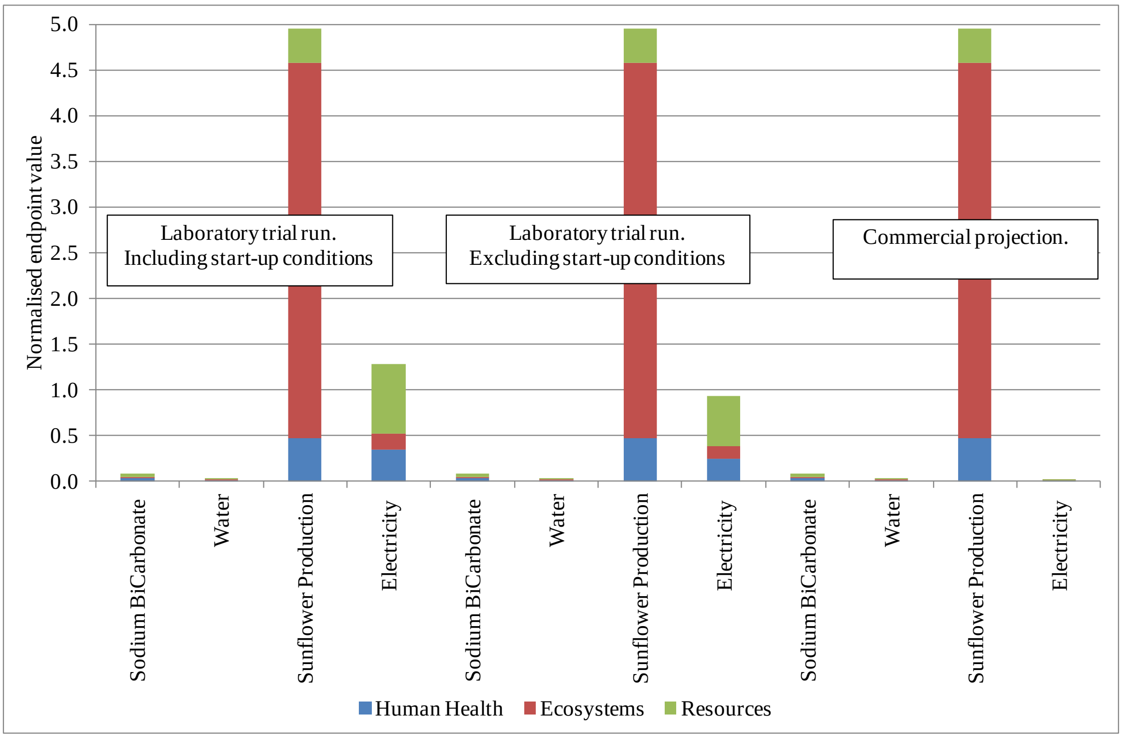

Figure E 7-1:Sunflowerseed WOB production – laboratory trials vs commercial projection: Normalised endpoint per area of protection (AoP) (including cultivation).

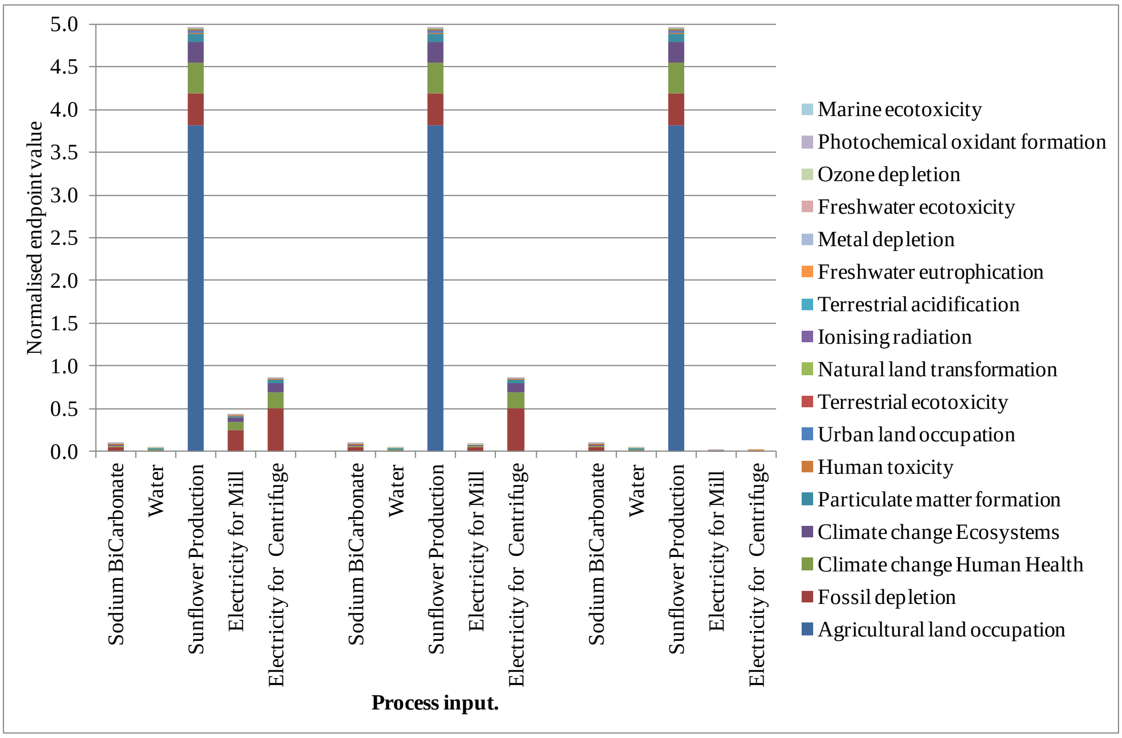

Figure E 7-2: Sunflowerseed WOB production – laboratory trials vs commercial projection: Normalised endpoint per impact category (including cultivation).

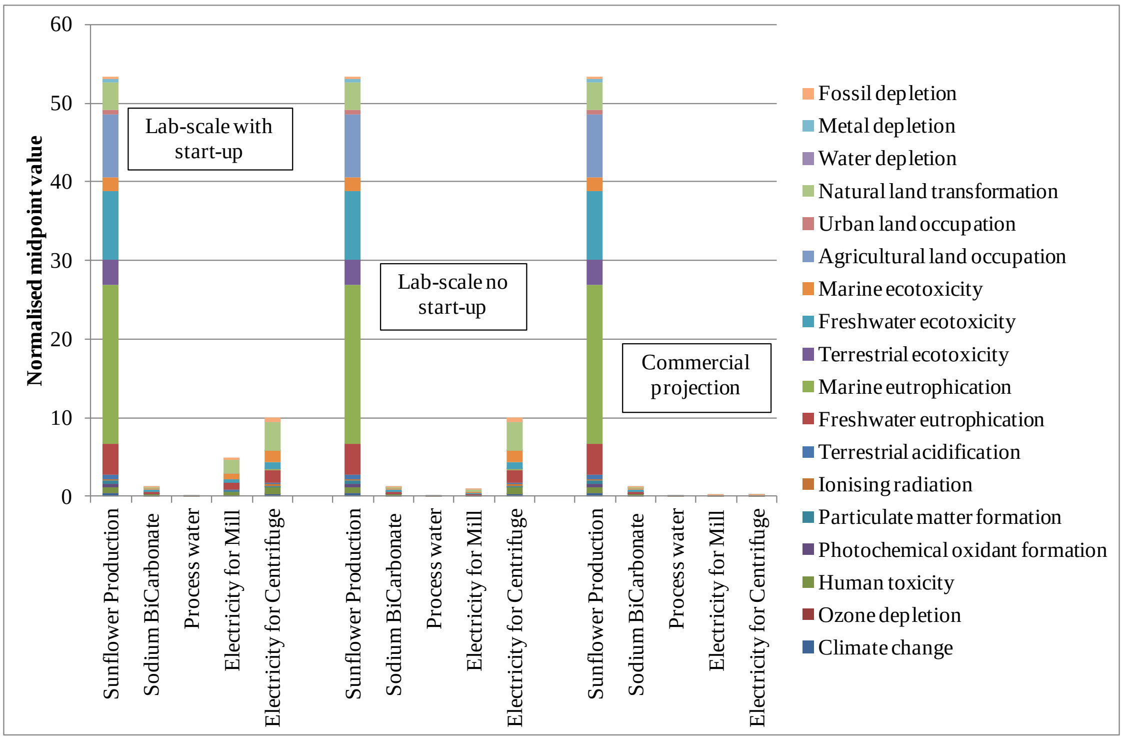

Figure E 7-3: Sunflowerseed WOB production – laboratory trials vs commercial projection: Normalised midpoint chart (including cultivation).

MIDPOINT	Lab-scale including start-up power				Lab-scale excluding start-up power				Commercial-scale projection.			
	Sodium Bi Carbon ate	Process water	Electricity for Mill	Electricity for Centrifuge	Sodium Bi Carbonate	Process water	Electricity for Mill	Electricity for Centrifuge	Sodium Bi Carbonate	Process water	Electricity for Mill	Electricity for Centrifuge
Climate change	0.0243	0.0107	0.1143	0.2303	0.0243	0.0107	0.0211	0.2303	0.0243	0.0107	0.0014	0.0022
Ozone depletion	0.0005	0.0006	0.0016	0.0033	0.0005	0.0006	0.0003	0.0033	0.0005	0.0006	0.0000	0.0000
Human toxicity	0.1785	0.0039	0.4155	0.8373	0.1785	0.0039	0.0768	0.8373	0.1785	0.0039	0.0050	0.0080
Photochemical oxidant formation	0.0121	0.0047	0.0470	0.0947	0.0121	0.0047	0.0087	0.0947	0.0121	0.0047	0.0006	0.0009
Particulate matter formation	0.0320	0.0068	0.0797	0.1606	0.0320	0.0068	0.0147	0.1606	0.0320	0.0068	0.0010	0.0015
Ionising radiation	0.0072	0.0024	0.0912	0.1839	0.0072	0.0024	0.0169	0.1839	0.0072	0.0024	0.0011	0.0017
Terrestrial acidification	0.0509	0.0092	0.1111	0.2239	0.0509	0.0092	0.0205	0.2239	0.0509	0.0092	0.0013	0.0021
Freshwater eutrophication	0.2484	0.0006	0.8232	1.6588	0.2484	0.0006	0.1522	1.6588	0.2484	0.0006	0.0100	0.0158
Marine eutrophication	0.0136	0.0022	0.0196	0.0396	0.0136	0.0022	0.0036	0.0396	0.0136	0.0022	0.0002	0.0004
Terrestrial ecotoxicity	0.0055	0.0001	0.0026	0.0052	0.0055	0.0001	0.0005	0.0052	0.0055	0.0001	0.0000	0.0000
Freshwater ecotoxicity	0.2085	0.0019	0.4789	0.9651	0.2085	0.0019	0.0885	0.9651	0.2085	0.0019	0.0058	0.0092
Marine ecotoxicity	0.2217	0.0018	0.6684	1.3470	0.2217	0.0018	0.1236	1.3470	0.2217	0.0018	0.0081	0.0128
Agricultural land occupation	0.0025	0.0000	0.0035	0.0071	0.0025	0.0000	0.0007	0.0071	0.0025	0.0000	0.0000	0.0001
Urban land occupation	0.0034	0.0000	0.0128	0.0258	0.0034	0.0000	0.0024	0.0258	0.0034	0.0000	0.0002	0.0002
Natural land transformation	0.1669	0.0000	1.8048	3.6371	0.1669	0.0000	0.3336	3.6371	0.1669	0.0000	0.0219	0.0346
Water depletion	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Metal depletion	0.0270	0.0003	0.0111	0.0224	0.0270	0.0003	0.0021	0.0224	0.0270	0.0003	0.0001	0.0002
Fossil depletion	0.0435	0.0159	0.2506	0.5051	0.0435	0.0159	0.0463	0.5051	0.0435	0.0159	0.0030	0.0048

Table E7-1: Sunflowerseed WOB production – laboratory trials vs commercial projection: Normalised midpoint data.

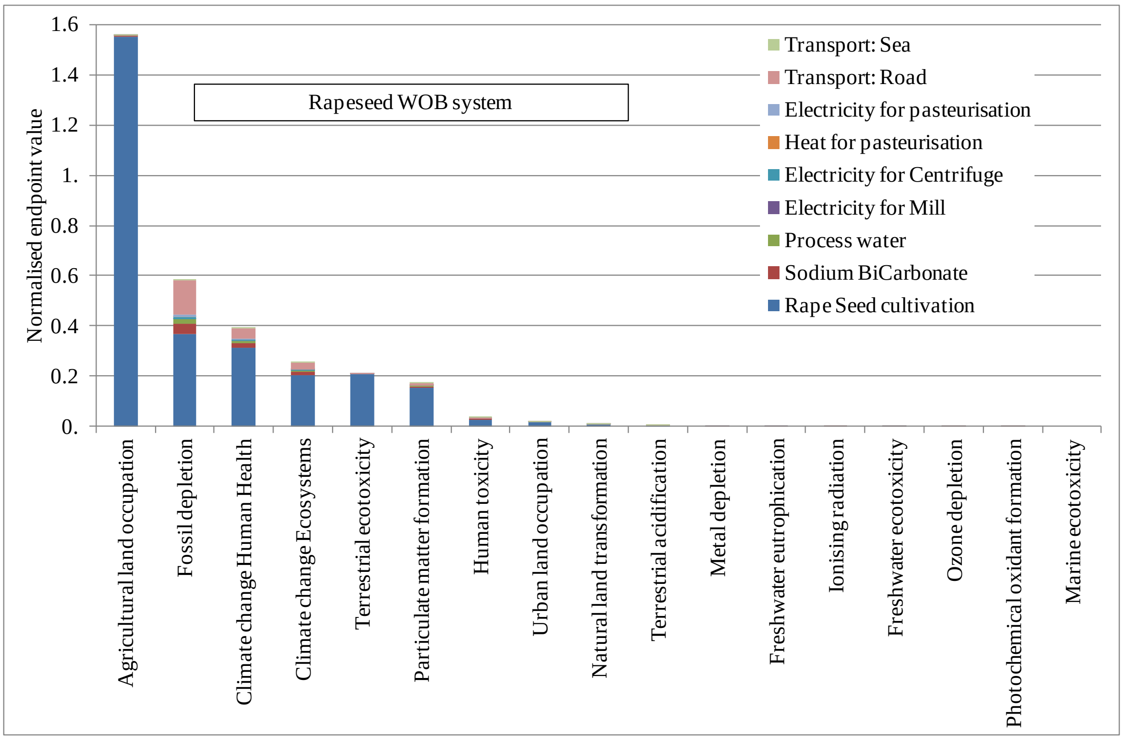

Figure E 7-4: Normalised endpoint LCIA for Rapeseed WOB system, ranked in order of magnitude

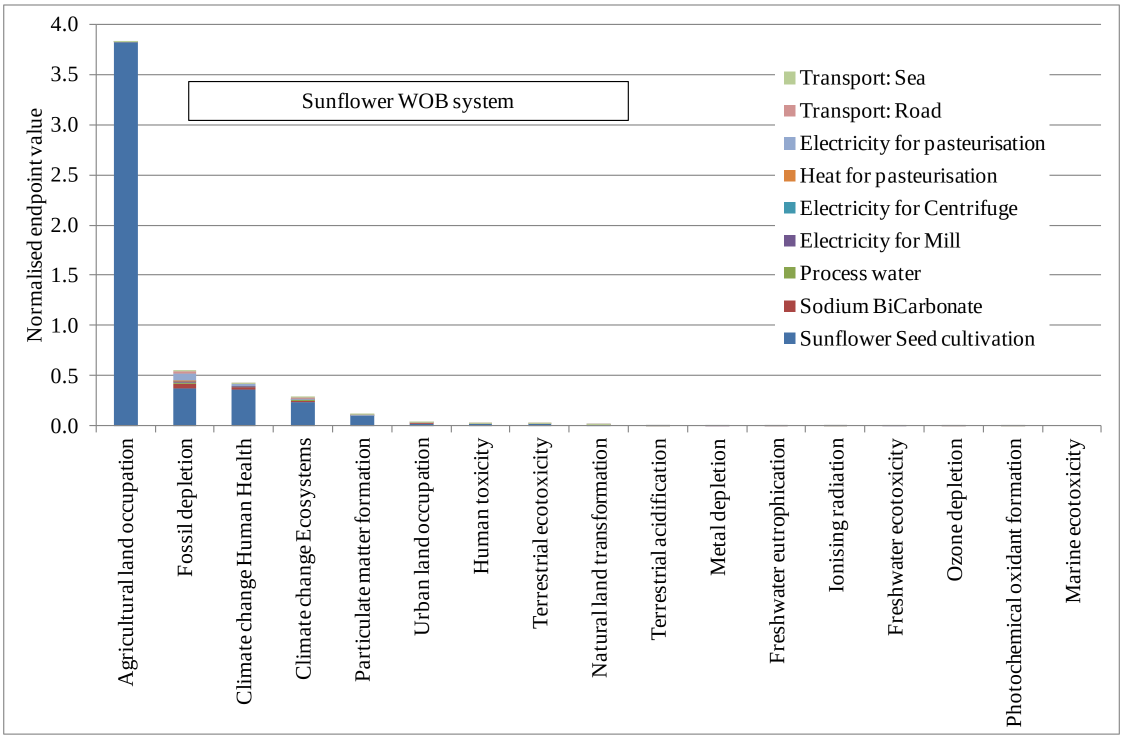

Figure E 7-5: Normalised endpoint LCIA for Rapeseed WOB system, ranked in order of magnitude.

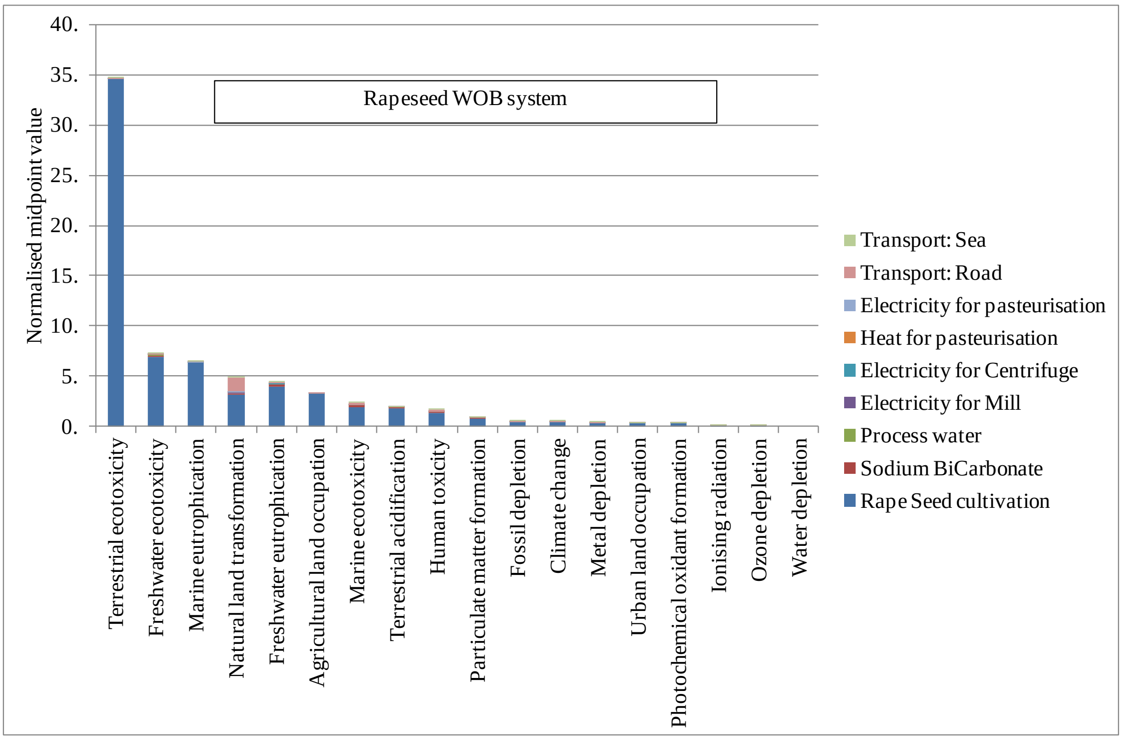

Figure E 7-6: Normalised midpoint LCIA for Rapeseed WOB system, ranked in order of magnitude.

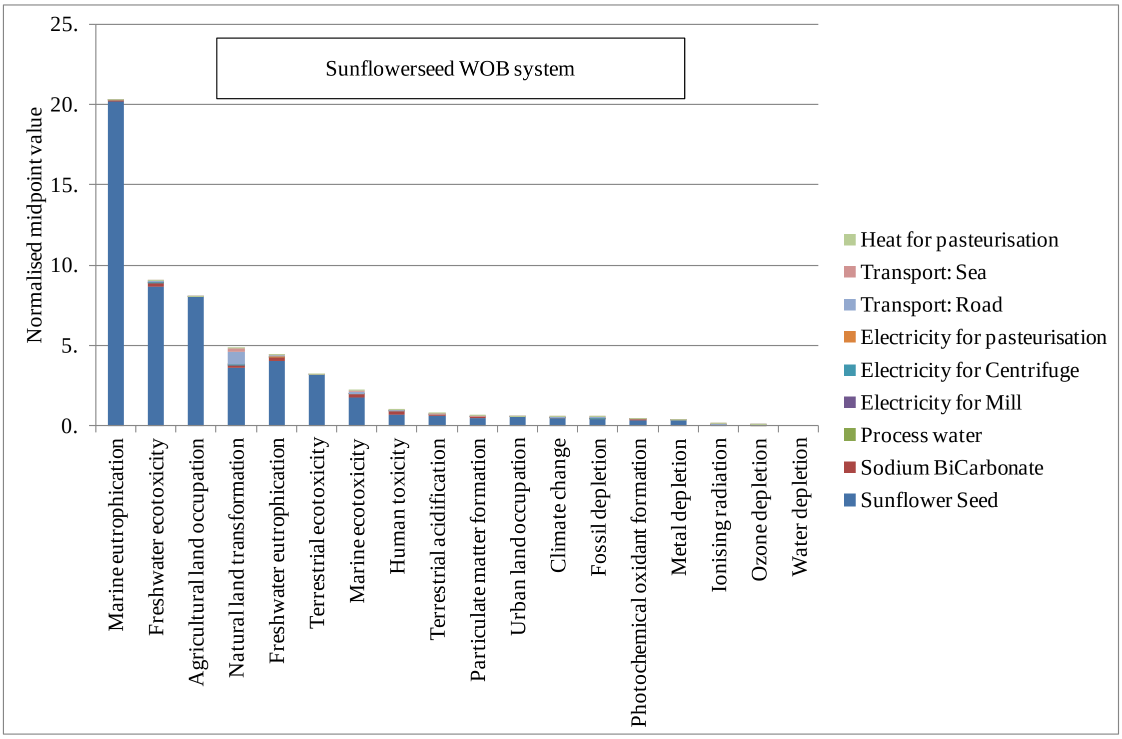

Figure E 7-7: Normalised midpoint LCIA for Sunflowerseed WOB system, ranked in order of magnitude.

			Rape seed cultivation	Sodium BiCarbonate	Water	Electricity for Mill	Electricity for Centrifuge	Electricity for pasteurisation	Heat for pasteurisation	Transport: Road	Transport: Sea	Transport: Road from farm
Climate change	kg CO2 eq	5598.25	4521.00	272.39	120.27	15.52	24.57	29.10	17.92	32.83	4.18	560.47
Ozone depletion	kg CFC-11 eq	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Human toxicity	kg 1,4-DB eq	951.14	758.18	105.72	2.32	2.98	4.72	5.59	1.46	3.85	0.59	65.74
Photochemical oxidant formation	kg NMVOC	18.56	14.70	0.64	0.25	0.03	0.05	0.06	0.03	0.15	0.06	2.59
Particulate matter formation	kg PM10 eq	13.35	11.88	0.48	0.10	0.01	0.02	0.03	0.01	0.04	0.03	0.75
Ionising radiation	kg U235 eq	475.90	327.61	44.88	14.81	6.91	10.93	12.95	1.10	3.11	0.55	53.05
Terrestrial acidification	kg SO2 eq	66.73	62.54	1.75	0.32	0.05	0.07	0.09	0.04	0.10	0.09	1.69
Freshwater eutrophication	kg P eq	1.82	1.64	0.10	0.00	0.00	0.01	0.01	0.00	0.00	0.00	0.05
Marine eutrophication	kg N eq	64.78	64.51	0.14	0.02	0.00	0.00	0.00	0.00	0.01	0.00	0.09
Terrestrial ecotoxicity	kg 1,4-DB eq	284.20	284.05	0.05	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.10
Freshwater ecotoxicity	kg 1,4-DB eq	78.79	74.69	2.27	0.02	0.06	0.10	0.12	0.03	0.08	0.01	1.41
Marine ecotoxicity	kg 1,4-DB eq	20.31	16.23	1.88	0.01	0.07	0.11	0.13	0.04	0.10	0.02	1.71
Agricultural land occupation	m2a	14782.17	14767.70	11.24	0.00	0.19	0.31	0.36	0.08	0.13	0.01	2.15
Urban land occupation	m2a	143.29	134.37	1.37	0.00	0.06	0.10	0.12	0.03	0.40	0.02	6.83
Natural land transformation	m2	0.77	0.51	0.03	0.00	0.00	0.01	0.01	0.00	0.01	0.00	0.20
Water depletion	m3	42.13	11.77	8.93	18.79	0.09	0.14	0.17	0.02	0.12	0.01	2.09
Metal depletion	kg Fe eq	264.51	216.67	19.24	0.20	0.10	0.15	0.18	0.07	1.54	0.06	26.30
Fossil depletion	kg oil eq	959.56	611.45	72.26	26.47	5.05	8.00	9.47	6.44	12.12	1.42	206.88

Table E7-2: Rapeseed oil-body production system: Characterised midpoints, ReCiPe(2008).

Impact category	Unit	Sunflower cultivation	Sodium Bi Carbonate	Process water	Electricity for Mill	Electricity for Centrifuge	Electricity for past - eurisation	Transport: Road	Transport: Sea	Heat for pasteuris - ation	Electricity for past - eurisation (25 kWh)
Climate change	kg CO2 eq	85.3%	4.5%	2.0%	0.3%	0.4%	0.5%	5.3%	1.5%	0.3%	0.2%
Ozone depletion	kg CFC-11 eq	77.3%	2.7%	3.2%	0.1%	0.2%	0.2%	13.2%	2.6%	0.5%	0.1%
Human toxicity	kg 1,4-DB eq	71.1%	17.6%	0.4%	0.5%	0.8%	0.9%	6.4%	2.1%	0.2%	0.4%
Photochemical oxidant formation	kg NMVOC	83.0%	2.8%	1.1%	0.1%	0.2%	0.2%	6.5%	5.9%	0.1%	0.1%
Particulate matter formation	kg PM10 eq	81.6%	5.3%	1.1%	0.2%	0.3%	0.3%	4.9%	6.3%	0.1%	0.1%
Ionising radiation	kg U235 eq	75.8%	8.1%	2.7%	1.2%	2.0%	2.3%	5.6%	2.1%	0.2%	1.1%
Terrestrial acidification	kg SO2 eq	80.5%	6.7%	1.2%	0.2%	0.3%	0.3%	3.8%	6.9%	0.2%	0.2%
Freshwater eutrophication	kg P eq	90.8%	5.6%	0.0%	0.2%	0.4%	0.4%	1.6%	0.8%	0.1%	0.2%
Marine eutrophication	kg N eq	99.9%	0.1%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
Terrestrial ecotoxicity	kg 1,4-DB eq	99.6%	0.2%	0.0%	0.0%	0.0%	0.0%	0.2%	0.0%	0.0%	0.0%
Freshwater ecotoxicity	kg 1,4-DB eq	96.2%	2.3%	0.0%	0.1%	0.1%	0.1%	0.8%	0.3%	0.0%	0.1%
Marine ecotoxicity	kg 1,4-DB eq	79.7%	10.2%	0.1%	0.4%	0.6%	0.7%	5.4%	2.7%	0.2%	0.3%
Agricultural land occupation	m2a	100.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
Urban land occupation	m2a	97.4%	0.6%	0.0%	0.0%	0.0%	0.1%	1.7%	0.2%	0.0%	0.0%
Natural land transformation	m2	73.6%	3.4%	0.0%	0.4%	0.7%	0.8%	15.0%	5.3%	0.6%	0.4%
Water depletion	m3	23.0%	23.3%	48.9%	0.2%	0.4%	0.4%	3.2%	0.5%	0.0%	0.2%
Metal depletion	kg Fe eq	85.4%	7.7%	0.1%	0.0%	0.1%	0.1%	6.1%	0.5%	0.0%	0.0%
Fossil depletion	kg oil eq	69.2%	8.0%	2.9%	0.6%	0.9%	1.0%	13.3%	3.3%	0.7%	0.5%

Table E7-3: Percentage contributions to midpoint values – Sunflowerseed oil-body production, with data for reduced pasteurisation power indicated.

Impact category	Unit	Rapeseed cultivation	Sodium Bi Carbonate	Process water	Electricity for Mill	Electricity for Centrifuge	Electricity for past - eurisation (54 kWh)	Transport: Sea	Transport: Road	Heat for pasteuris - ation	Electricity for past - eurisation (25 kWh)
Climate change	kg CO2 eq	80.3%	4.8%	2.1%	0.3%	0.4%	0.3%	0.5%	11.2%	0.0%	0.2%
Ozone depletion	kg CFC-11 eq	74.2%	2.2%	2.5%	0.1%	0.1%	0.4%	0.2%	20.2%	0.0%	0.1%
Human toxicity	kg 1,4-DB eq	79.4%	11.1%	0.2%	0.3%	0.5%	0.2%	0.6%	7.7%	0.0%	0.3%
Photochemical oxidant formation	kg NMVOC	78.6%	3.4%	1.3%	0.2%	0.3%	0.1%	0.3%	15.6%	0.2%	0.1%
Particulate matter formation	kg PM10 eq	88.7%	3.6%	0.8%	0.1%	0.2%	0.1%	0.2%	6.3%	0.1%	0.1%
Ionising radiation	kg U235 eq	68.4%	9.4%	3.1%	1.4%	2.3%	0.2%	2.7%	12.5%	0.1%	1.3%
Terrestrial acidification	kg SO2 eq	93.6%	2.6%	0.5%	0.1%	0.1%	0.1%	0.1%	2.8%	0.1%	0.1%
Freshwater eutrophication	kg P eq	90.1%	5.7%	0.0%	0.2%	0.4%	0.1%	0.4%	3.1%	0.0%	0.2%
Marine eutrophication	kg N eq	99.6%	0.2%	0.0%	0.0%	0.0%	0.0%	0.0%	0.2%	0.0%	0.0%
Terrestrial ecotoxicity	kg 1,4-DB eq	99.9%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
Freshwater ecotoxicity	kg 1,4-DB eq	94.7%	2.9%	0.0%	0.1%	0.1%	0.0%	0.1%	2.0%	0.0%	0.1%
Marine ecotoxicity	kg 1,4-DB eq	79.5%	9.2%	0.1%	0.3%	0.5%	0.2%	0.6%	9.4%	0.1%	0.3%
Agricultural land occupation	m2a	99.9%	0.1%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
Urban land occupation	m2a	93.5%	1.0%	0.0%	0.0%	0.1%	0.0%	0.1%	5.3%	0.0%	0.0%
Natural land transformation	m2	64.7%	3.4%	0.0%	0.5%	0.7%	0.6%	0.8%	29.1%	0.1%	0.4%
Water depletion	m3	27.8%	21.1%	44.5%	0.2%	0.3%	0.0%	0.4%	5.6%	0.0%	0.2%
Metal depletion	kg Fe eq	81.4%	7.2%	0.1%	0.0%	0.1%	0.0%	0.1%	11.1%	0.0%	0.0%
Fossil depletion	kg oil eq	62.9%	7.4%	2.7%	0.5%	0.8%	0.7%	1.0%	23.9%	0.1%	0.5%

Table E7-4: Percentage contributions to midpoint values – Rapeseed oil-body production, with data for reduced pasteurisation power indicated.

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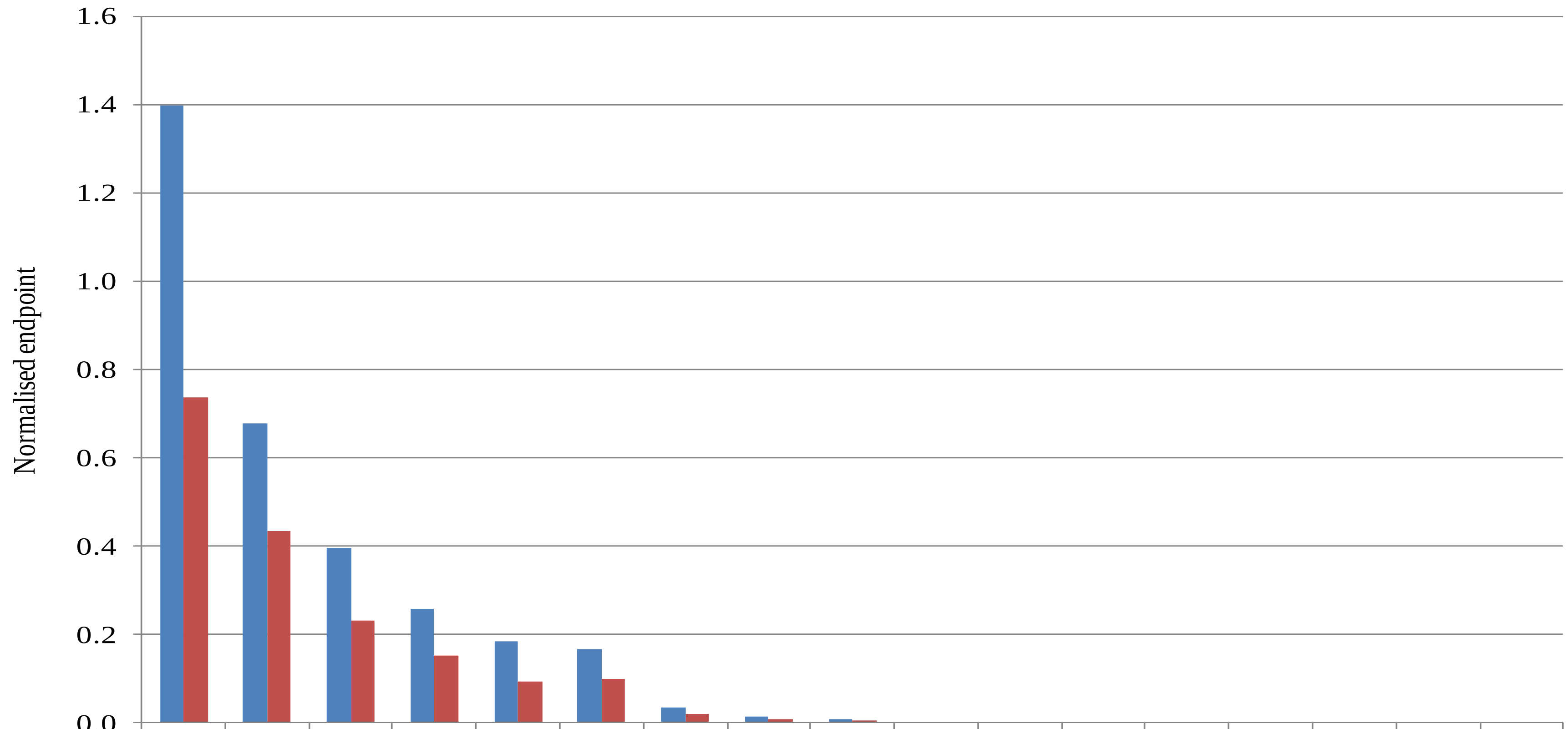
APPENDIX F. **OIL-BODY MAYONNAISE LCA**

Chapter 8 - Supplementary material

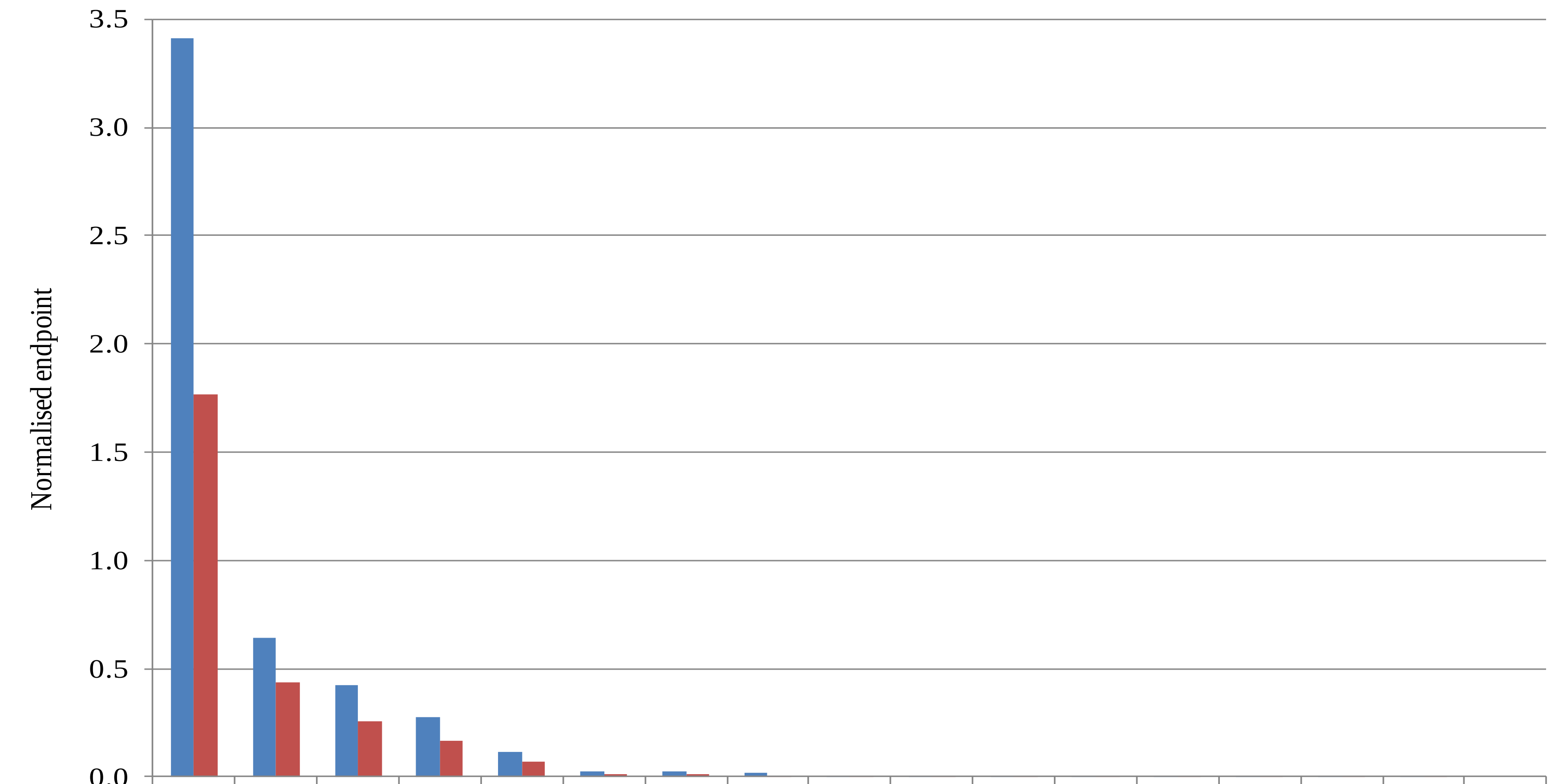
GLOSSARY OF TERMS

ACRONYM	FULL TITLE
RS	Rapeseed
RSM	Rapeseed mayonnaise
RSOBM	Rapeseed oil-body mayonnaise
SF	Sunflower
SFM	Sunflower mayonnaise
SFOBM	Sunflower oil-body mayonnaise
Impact categories	
ALO	Agricultural land occupation
CC	Climate change
FD	Fossil depletion
FET	Freshwater ecotoxicity
FE	Freshwater eutrophication
HT	Human toxicity
IR	Ionising radiation
MET	Marine ecotoxicity
ME	Marine eutrophication
MD	Metal depletion
NLT	Natural land transformation
OD	Ozone depletion
PMF	Particulate matter formation
POF	Photochemical oxidant formation
TA	Terrestrial acidification
TET	Terrestrial ecotoxicity
ULO	Urban land occupation
WD	Water depletion



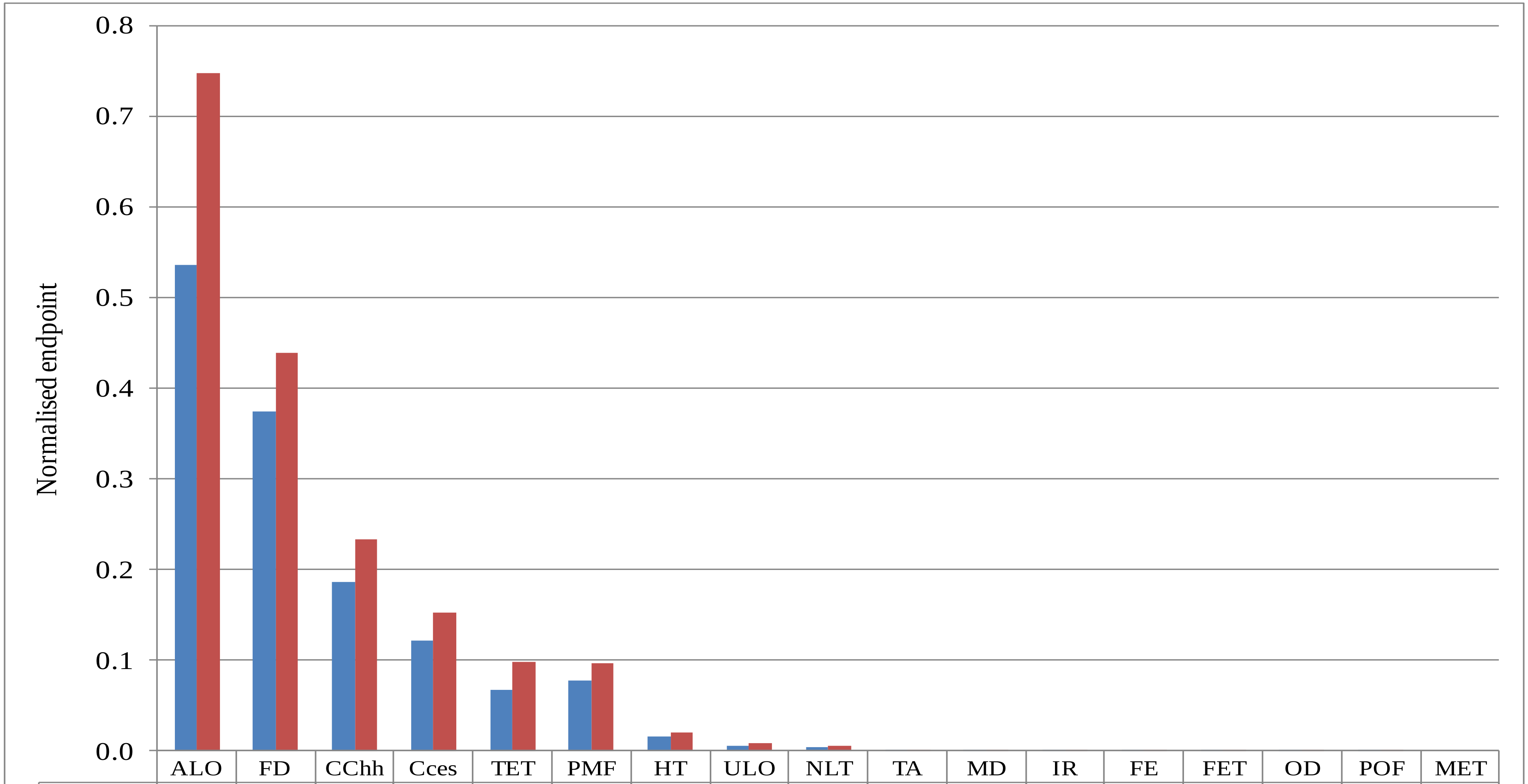
	ALO	FD	CCHH	CCES	TET	PMF	HT	ULO	NLT	TA	MD	FE	IR	FET	OD	POF	ME
RSOBM	1.3989	0.6773	0.3952	0.2584	0.1848	0.1678	0.0347	0.0147	0.0095	0.0021	0.0007	0.0004	0.0004	0.0001	0.0001	0.0000	0.0000
RSM	0.7363	0.4346	0.2313	0.1512	0.0943	0.0981	0.0192	0.0077	0.0055	0.0013	0.0003	0.0002	0.0002	0.0001	0.0000	0.0000	0.0000

Figure F 8-1: Normalised endpoint results for UNALLOCATED rapeseed mayonnaise and OBM.



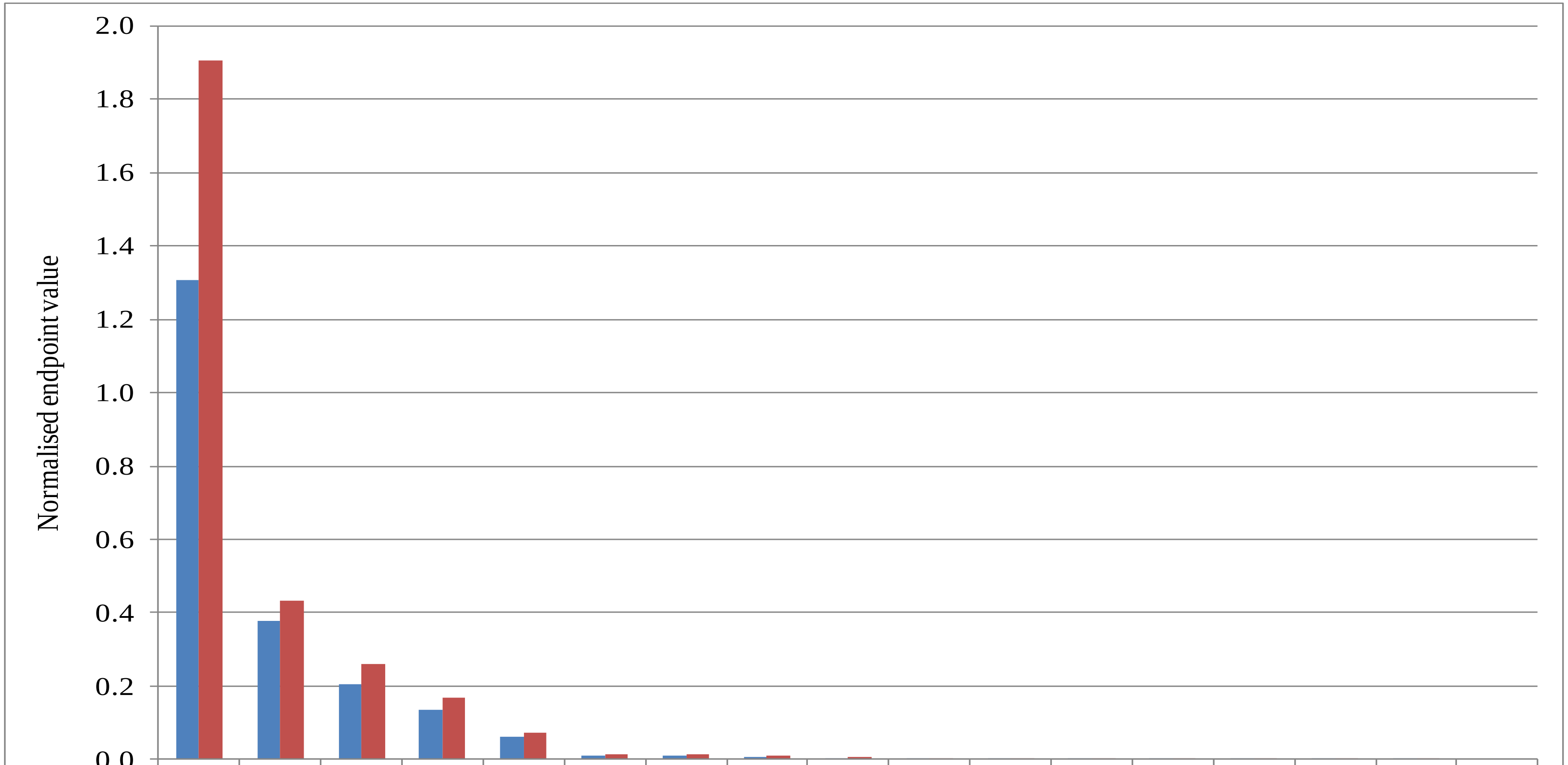
	ALO	FD	CCHH	CCES	PMF	HT	ULO	TET	NLT	TA	MD	IR	FE	FET	OD	POF	MET
SFOBM	3.4129	0.6411	0.4247	0.2776	0.1171	0.0238	0.0234	0.0170	0.0085	0.0009	0.0006	0.0005	0.0004	0.0001	0.0001	0.0000	0.0000
SFM	1.7645	0.4381	0.2531	0.1655	0.0723	0.0141	0.0122	0.0087	0.0053	0.0006	0.0003	0.0003	0.0002	0.0001	0.0000	0.0000	0.0000

Figure F 8-2: Normalised endpoint results for UNALLOCATED sunflowerseed mayonnaise and OBM.



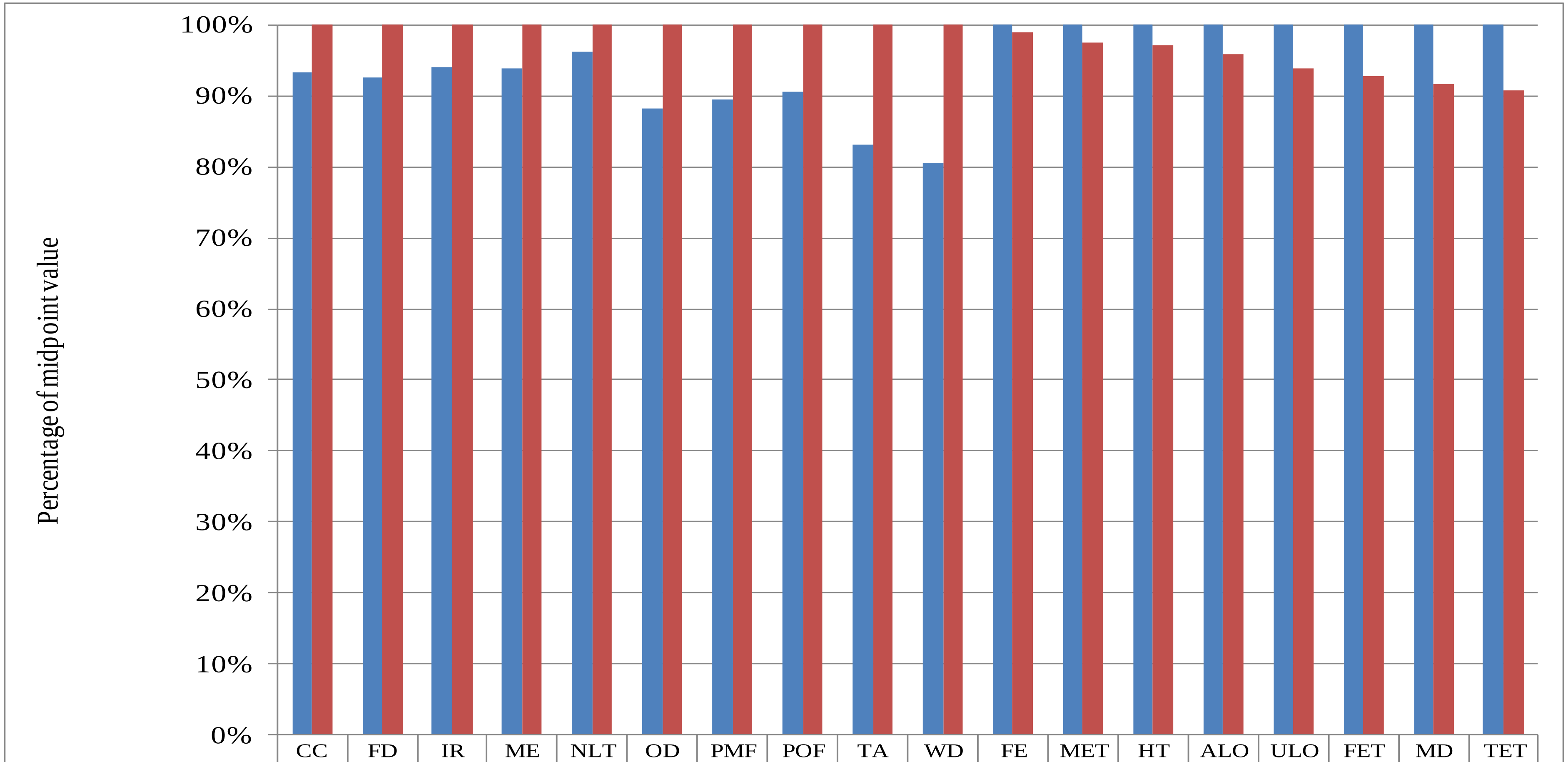
	ALO	FD	CChh	Cces	TET	PMF	HT	ULO	NLT	TA	MD	IR	FE	FET	OD	POF	MET
RSM	0.5368	0.3746	0.1870	0.1223	0.0676	0.0778	0.0157	0.0057	0.0046	0.0010	0.0003	0.0002	0.0002	0.0000	0.0000	0.0000	0.0000
RSOBM	0.7480	0.4397	0.2335	0.1527	0.0978	0.0961	0.0209	0.0081	0.0059	0.0012	0.0004	0.0002	0.0002	0.0001	0.0000	0.0000	0.0000

Figure F 8-3: Normalised endpoint values for rapeseed mayonnaise and OBM with ECONOMIC ALLOCATION.



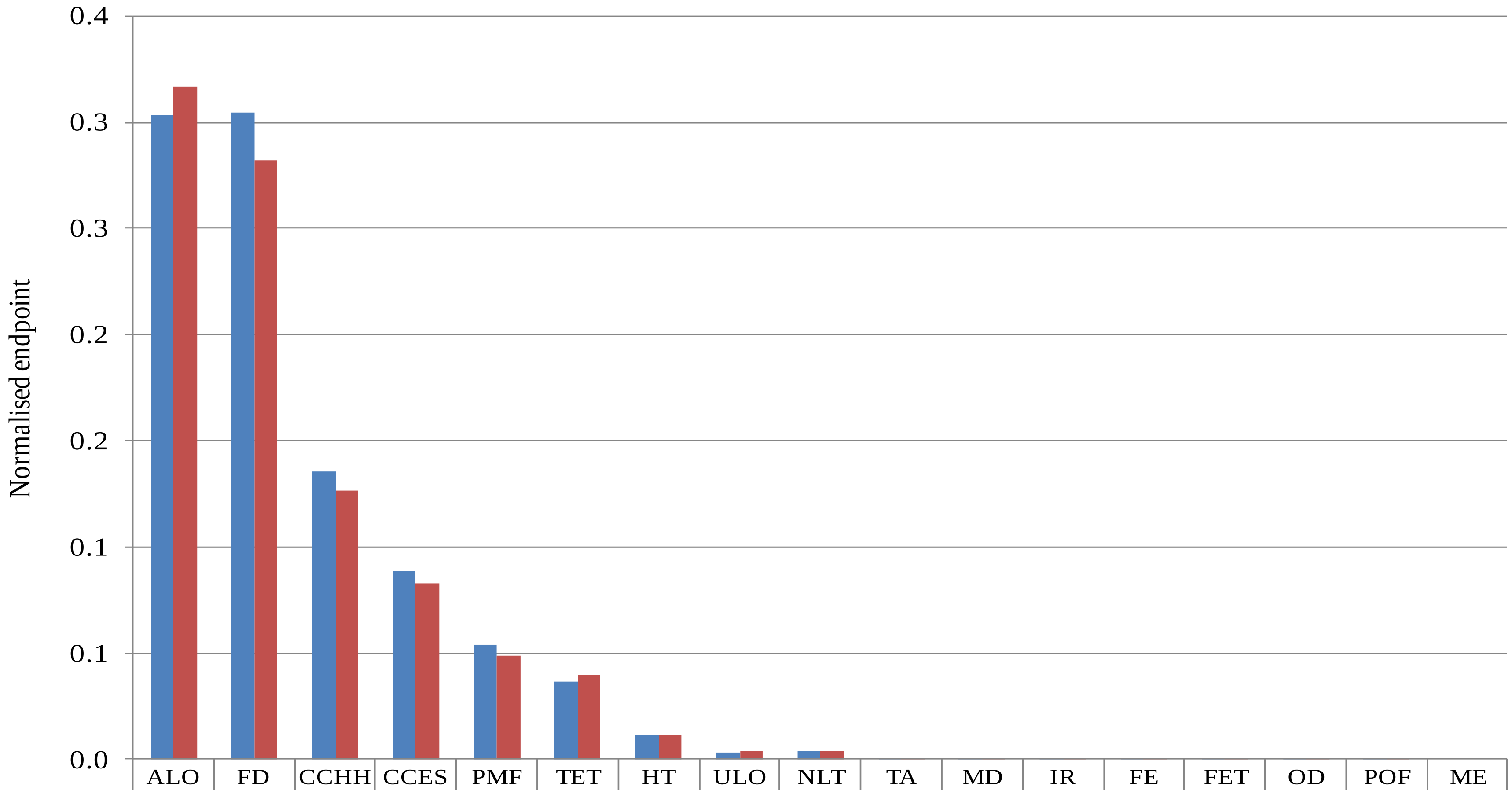
	ALO	FD	CCHH	CCES	PMF	HT	ULO	TET	NLT	TA	MD	IR	FE	FET	OD	POF	MET
SFM	1.3090	0.3778	0.2052	0.1341	0.0603	0.0120	0.0092	0.0064	0.0044	0.0005	0.0003	0.0002	0.0002	0.0001	0.0000	0.0000	0.0000
SFOBM	1.9050	0.4333	0.2592	0.1695	0.0720	0.0157	0.0133	0.0095	0.0055	0.0006	0.0004	0.0003	0.0003	0.0001	0.0000	0.0000	0.0000

Figure F 8-4: Normalised endpoint values for sunflowerseed mayonnaise and OBM with ECONOMIC ALLOCATION.



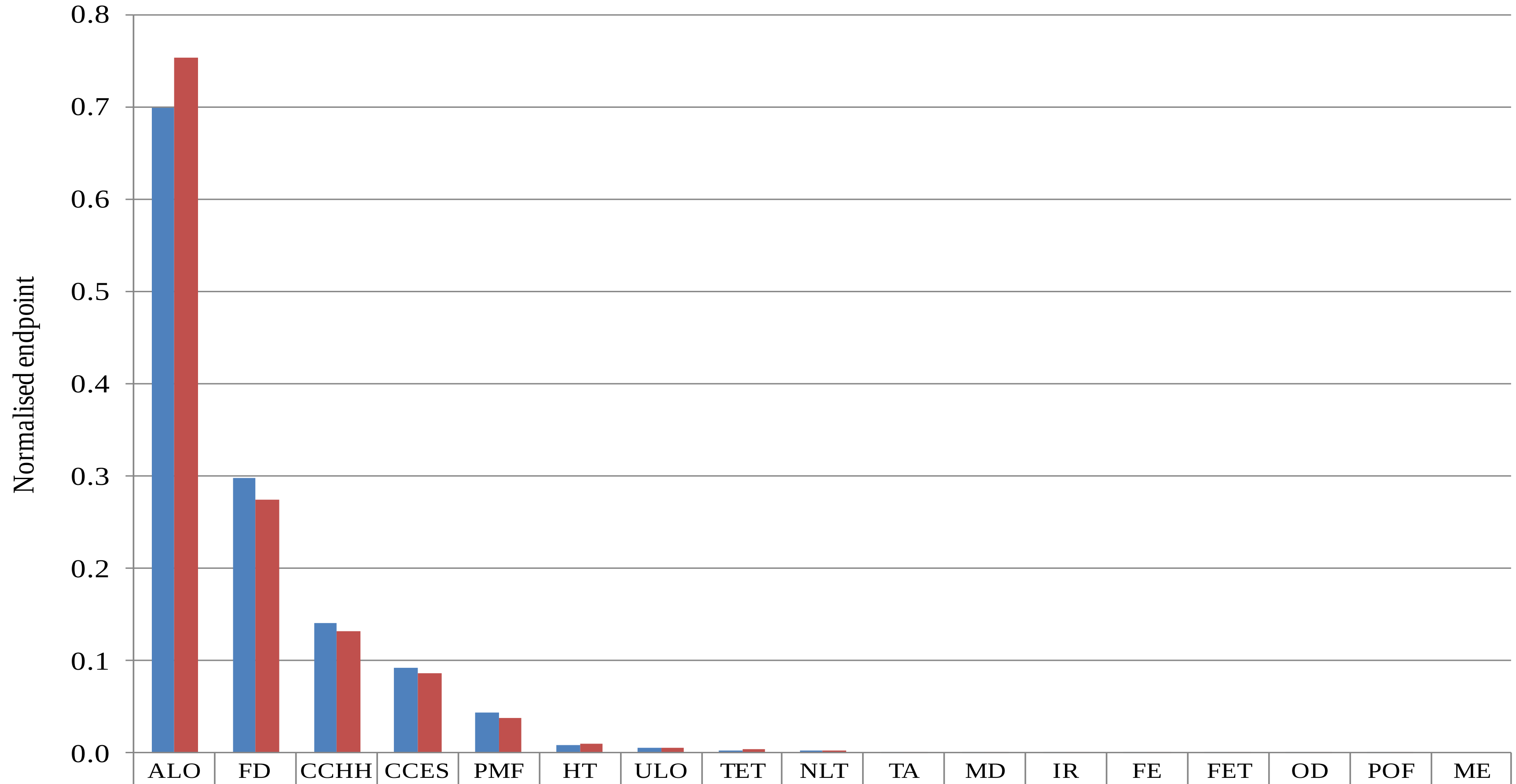
	CC	FD	IR	ME	NLT	OD	PMF	POF	TA	WD	FE	MET	HT	ALO	ULO	FET	MD	TET
Rapeseed OBM	93.4%	92.7%	94.1%	93.8%	96.3%	88.3%	89.6%	90.6%	83.1%	80.6%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%
Rapeseed mayonnaise	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	98.9%	97.6%	97.2%	96.0%	93.8%	92.9%	91.8%	90.9%

Figure F 8-5: Percentage of total characterised midpoint values for rapeseed mayonnaise and OBM - MASS ALLOCATION.



	ALO	FD	CCHH	CCES	PMF	TET	HT	ULO	NLT	TA	MD	IR	FE	FET	OD	POF	ME
RSM	0.3036	0.3045	0.1353	0.0885	0.0542	0.0365	0.0115	0.0035	0.0036	0.0007	0.0002	0.0002	0.0001	0.0000	0.0000	0.0000	0.0000
RSOBM	0.3167	0.2823	0.1264	0.0827	0.0486	0.0401	0.0118	0.0037	0.0035	0.0006	0.0002	0.0001	0.0001	0.0000	0.0000	0.0000	0.0000

Figure F 8-6: Normalised endpoint values for rapeseed mayonnaise and OBM with MASS ALLOCATION.



	ALO	FD	CCHH	CCES	PMF	HT	ULO	TET	NLT	TA	MD	IR	FE	FET	OD	POF	ME
SFM	0.6999	0.2974	0.1411	0.0923	0.0442	0.0094	0.0052	0.0034	0.0033	0.0004	0.0002	0.0002	0.0001	0.0000	0.0000	0.0000	0.0000
SFOBM	0.7531	0.2745	0.1328	0.0868	0.0376	0.0095	0.0056	0.0038	0.0033	0.0003	0.0002	0.0002	0.0001	0.0000	0.0000	0.0000	0.0000

Figure F 8-7: Normalised endpoint values for sunflowerseed mayonnaise and OBM with MASS ALLOCATION

Impact category	Agricultural land occupation		Fossil depletion		Climate change		Terrestrial ecotoxicity	
Unit	m^2a	% Contribution	kg oil eq	% Contribution	kg CO_2 eq	% Contribution	kg 1,4-DB eq	% Contribution
Total	13409.6		1129.0		5693		252.7337	
Bicarbonate	10.0	0.1%	64.2	5.7%	242	4.2%	0.0401	0.02%
Board	9.5	0.1%	1.5	0.1%	5	0.1%	0.0013	0.00%
Centrifuge power	0.3	0.0%	7.1	0.6%	22	0.4%	0.0004	0.00%
Film	0.8	0.0%	9.4	0.8%	13	0.2%	0.0004	0.00%
Gas (fuel) – food plant	0.2	0.0%	101.7	9.0%	260	4.6%	0.0044	0.00%
Glass	118.4	0.9%	152.2	13.5%	419	7.4%	0.1232	0.05%
Mill power	0.2	0.0%	4.5	0.4%	14	0.2%	0.0002	0.00%
Pallet	142.1	1.1%	5.2	0.5%	11	0.2%	0.0049	0.00%
Pasteurisation heat	0.1	0.0%	5.7	0.5%	16	0.3%	0.0017	0.00%
Pasteurisation power	0.3	0.0%	8.4	0.7%	26	0.5%	0.0004	0.00%
Road - from farm	1.9	0.0%	183.7	16.3%	498	8.7%	0.0853	0.03%
Road transport	0.1	0.0%	10.8	1.0%	29	0.5%	0.0050	0.00%
Salt	0.1	0.0%	0.3	0.0%	1	0.0%	0.0001	0.00%
Sea - from farm		0.0%						
Sea transport	0.0	0.0%	1.3	0.1%	4	0.1%	0.0003	0.00%
Seed	13113.7	97.8%	543.0	48.1%	4015	70.5%	252.2350	99.80%
Vinegar for mayonnaise	0.1	0.0%	6.0	0.5%	8	0.1%	0.0026	0.00%
Water	0.0	0.0%	23.5	2.1%	107	1.9%	0.0011	0.00%
Water	0.0	0.0%	0.0	0.0%	0	0.0%	0.0000	0.00%

Table F 8-1: Process contributions to top four impact categories: Rapeseed OBM - No allocation.

Impact category	Agricultural land occupation		Fossil depletion		Climate change		Particulate matter formation	
Unit	m^2a	% Contribution	kg oil eq	% Contribution	kg CO_2 eq	% Contribution	kg PM10 eq	% Contribution
Total	32525.2		1069.0		6118		9.1	
Bicarbonate	10.0	0.1%	64.2	6.0%	242	4.0%	0.42	4.7%
Board	9.5	0.1%	1.5	0.1%	5	0.1%	0.01	0.1%
Centrifuge power	0.3	0.0%	7.1	0.7%	22	0.4%	0.02	0.2%
Film	0.8	0.0%	9.4	0.9%	13	0.2%	0.02	0.2%
Gas (fuel) - food plant	0.2	0.0%	101.7	9.5%	260	4.3%	0.07	0.8%
Glass	118.4	0.9%	152.2	14.2%	419	6.8%	1.02	11.2%
Mill power	0.2	0.0%	4.5	0.4%	14	0.2%	0.01	0.1%
Pallet	142.1	1.1%	5.2	0.5%	11	0.2%	0.02	0.2%
Pasteurisation heat	0.1	0.0%	5.7	0.5%	16	0.3%	0.01	0.1%
Pasteurisation power	0.3	0.0%	8.4	0.8%	26	0.4%	0.02	0.3%
Road - from farm	0.9	0.0%	85.7	8.0%	232	3.8%	0.31	3.4%
Road transport	0.1	0.0%	10.8	1.0%	29	0.5%	0.04	0.4%
Salt	0.1	0.0%	0.3	0.0%	1	0.0%	0.00	0.0%
Sea - from farm	0.2	0.0%	0.0%	2.5%	77	1.3%	0.49	5.4%
Sea transport	0.0	0.0%	1.3	0.1%	4	0.1%	0.02	0.3%
Seed	32212.9	97.8%	554.9	51.9%	4628	75.6%	6.49	71.4%
Vinegar for mayonnaise	0.1	0.0%	6.0	0.6%	8	0.1%	0.01	0.1%
Water	0.0	0.0%	23.5	2.2%	107	1.7%	0.09	1.0%
Water	0.0	0.0%	0.0	0.0%	0	0.0%	0.00	0.0%

Table F 8-2: Process contributions to top four impact categories: Sunflowerseed OBM - No allocation.

Impact category	Agricultural land occupation		Fossil depletion		Climate change		Terrestrial ecotoxicity	
Unit	m^2a	% Contribution	kg oil eq	% Contribution	kg CO_2 eq	% Contribution	kg 1,4-DB eq	% Contribution
Total	7219.39		732.85		3364		133.72	
Bicarbonate	5.28	0.1%	33.96	4.6%	128	3.8%	0.02	0.02%
Board	9.53	0.1%	1.51	0.2%	5	0.1%	0.00	0.00%
Centrifuge power	0.14	0.0%	3.76	0.5%	12	0.3%	0.00	0.00%
Film	0.83	0.0%	9.38	1.3%	13	0.4%	0.00	0.00%
Gas (fuel) – food plant	0.18	0.0%	101.69	13.9%	260	7.7%	0.00	0.00%
Glass	118.39	1.6%	152.18	20.8%	419	12.4%	0.12	0.09%
Mill power	0.09	0.0%	2.37	0.3%	7	0.2%	0.00	0.00%
Pallet	142.09	2.0%	5.17	0.7%	11	0.3%	0.00	0.00%
Pasteurisation heat	0.04	0.0%	3.03	0.4%	8	0.3%	0.00	0.00%
Pasteurisation power	0.17	0.0%	4.45	0.6%	14	0.4%	0.00	0.00%
Road - from farm	1.01	0.0%	97.24	13.3%	263	7.8%	0.05	0.03%
Road transport	0.11	0.0%	10.77	1.5%	29	0.9%	0.01	0.00%
Salt	0.07	0.0%	0.25	0.0%	1	0.0%	0.00	0.00%
Sea - from farm		0.0%		0.0%				0.00%
Sea transport	0.01	0.0%	1.26	0.2%	4	0.1%	0.00	0.00%
Seed	6941.28	96.1%	287.40	39.2%	2125	63.2%	133.51	99.84%
Vinegar for mayonnaise	0.15	0.0%	5.97	0.8%	8	0.3%	0.00	0.00%
Water	0.00	0.0%	12.44	1.7%	57	1.7%	0.00	0.00%
Water	0.00	0.0%	0.00	0.0%	0	0.0%	0.00	0.00%

Table F 8-3: Process contributions to top four impact categories: Rapeseed OBM - Economic allocation.

Impact category	Agricultural land occupation		Fossil depletion		Climate change		Particulate matter formation	
Unit	m^2a	% Contribution	kg oil eq	% Contribution	kg CO_2 eq	% Contribution	kg PM10 eq	% Contribution
Total	18201.6		722.2		3734		5.6	
Bicarbonate	5.6	0.0%	35.7	4.9%	135	3.6%	0.24	4.2%
Board	9.5	0.1%	1.5	0.2%	5	0.1%	0.01	0.1%
Centrifuge power	0.2	0.0%	4.0	0.5%	12	0.3%	0.01	0.2%
Film	0.8	0.0%	9.4	1.3%	13	0.4%	0.02	0.3%
Gas (fuel) - food plant	0.2	0.0%	101.7	14.1%	260	7.0%	0.07	1.3%
Glass	118.4	0.7%	152.2	21.1%	419	11.2%	1.02	18.3%
Mill power	0.1	0.0%	2.5	0.3%	8	0.2%	0.01	0.1%
Pallet	142.1	0.8%	5.2	0.7%	11	0.3%	0.02	0.4%
Pasteurisation heat	0.0	0.0%	3.2	0.4%	9	0.2%	0.01	0.1%
Pasteurisation power	0.2	0.0%	4.7	0.6%	14	0.4%	0.01	0.2%
Road - from farm	0.5	0.0%	47.7	6.6%	129	3.5%	0.17	3.1%
Road transport	0.1	0.0%	10.8	1.5%	29	0.8%	0.04	0.7%
Salt	0.1	0.0%	0.3	0.0%	1	0.0%	0.00	0.0%
Sea - from farm	0.1	0.0%	14.6	2.0%	43	1.1%	0.27	4.9%
Sea transport	0.0	0.0%	1.3	0.2%	4	0.1%	0.02	0.4%
Seed	17923.6	98.5%	308.7	42.7%	2575	69.0%	3.61	64.6%
Vinegar for mayonnaise	0.1	0.0%	6.0	0.8%	8	0.2%	0.01	0.2%
Water	0.0	0.0%	13.1	1.8%	59	1.6%	0.05	0.9%
Water	0.0	0.0%	0.0	0.0%	0	0.0%	0.00	0.0%

Table F 8-4: Process contributions to top four impact categories: Sunflowerseed OBM - Economic allocation.

Impact category	Agricultural land occupation		Fossil depletion		Climate change		Terrestrial ecotoxicity	
Unit	m^2a	% Contribution	kg oil eq	% Contribution	kg CO_2 eq	% Contribution	kg 1,4-DB eq	% Contribution
Total	3117.98		470.36		1821		54.87	
Bicarbonate	2.16	0.1%	13.92	3.0%	52	2.9%	0.01	0.02%
Board	9.53	0.3%	1.51	0.3%	5	0.3%	0.00	0.00%
Centrifuge power	0.06	0.0%	1.54	0.3%	5	0.3%	0.00	0.00%
Film	0.83	0.0%	9.38	2.0%	13	0.7%	0.00	0.00%
Gas (fuel) – food plant	0.18	0.0%	101.69	21.6%	260	14.3%	0.00	0.01%
Glass	118.39	3.8%	152.18	32.4%	419	23.0%	0.12	0.22%
Mill power	0.04	0.0%	0.97	0.2%	3	0.2%	0.00	0.00%
Pallet	142.09	4.6%	5.17	1.1%	11	0.6%	0.00	0.01%
Pasteurisation heat	0.02	0.0%	1.24	0.3%	3	0.2%	0.00	0.00%
Pasteurisation power	0.07	0.0%	1.82	0.4%	6	0.3%	0.00	0.00%
Road - from farm	0.41	0.0%	39.84	8.5%	108	5.9%	0.02	0.03%
Road transport	0.11	0.0%	10.77	2.3%	29	1.6%	0.01	0.01%
Salt	0.07	0.0%	0.25	0.1%	1	0.0%	0.00	0.00%
Sea - from farm								
Sea transport	0.01	0.0%	1.26	0.3%	4	0.2%	0.00	0.00%
Seed	2843.86	91.2%	117.75	25.0%	871	47.8%	54.70	99.69%
Vinegar for mayonnaise	0.15	0.0%	5.97	1.3%	8	0.5%	0.00	0.00%
Water	0.00	0.0%	5.10	1.1%	23	1.3%	0.00	0.00%
Water	0.00	0.0%	0.00	0.0%	0	0.0%	0.00	0.00%

Table F 8-5: Process contributions to top four impact categories: Rapeseed OBM - Mass allocation.

Impact category	Agricultural land occupation		Fossil depletion		Climate change		Particulate matter formation	
Unit	m^2a	% Contribution	kg oil eq	% Contribution	kg CO_2 eq	% Contribution	kg PM10 eq	% Contribution
Total	7259.7		457.4		1913		2.9	
Bicarbonate	2.2	0.0%	13.9	3.0%	52	2.7%	0.09	3.2%
Board	9.5	0.1%	1.5	0.3%	5	0.2%	0.01	0.2%
Centrifuge power	0.1	0.0%	1.5	0.3%	5	0.2%	0.00	0.2%
Film	0.8	0.0%	9.4	2.1%	13	0.7%	0.02	0.5%
Gas (fuel) - food plant	0.2	0.0%	101.7	22.2%	260	13.6%	0.07	2.5%
Glass	118.4	1.6%	152.2	33.3%	419	21.9%	1.02	35.0%
Mill power	0.0	0.0%	1.0	0.2%	3	0.2%	0.00	0.1%
Pallet	142.1	2.0%	5.2	1.1%	11	0.6%	0.02	0.7%
Pasteurisation heat	0.0	0.0%	1.2	0.3%	3	0.2%	0.00	0.1%
Pasteurisation power	0.1	0.0%	1.8	0.4%	6	0.3%	0.01	0.2%
Road - from farm	0.2	0.0%	18.6	4.1%	50	2.6%	0.07	2.3%
Road transport	0.1	0.0%	10.8	2.4%	29	1.5%	0.04	1.3%
Salt	0.1	0.0%	0.3	0.1%	1	0.0%	0.00	0.1%
Sea - from farm	0.0	0.0%	1.3	0.3%	4	0.2%	0.02	0.8%
Sea transport	0.0	0.0%	5.7	1.2%	17	0.9%	0.11	3.6%
Seed	6985.7	96.2%	120.3	26.3%	1004	52.5%	1.41	48.3%
Vinegar for mayonnaise	0.1	0.0%	6.0	1.3%	8	0.4%	0.01	0.3%
Water	0.0	0.0%	5.1	1.1%	23	1.2%	0.02	0.7%
Water	0.0	0.0%	0.0	0.0%	0	0.0%	0.00	0.0%

Table F8-6: Process contributions to top four impact categories: Sunflowerseed OBM - Mass allocation.

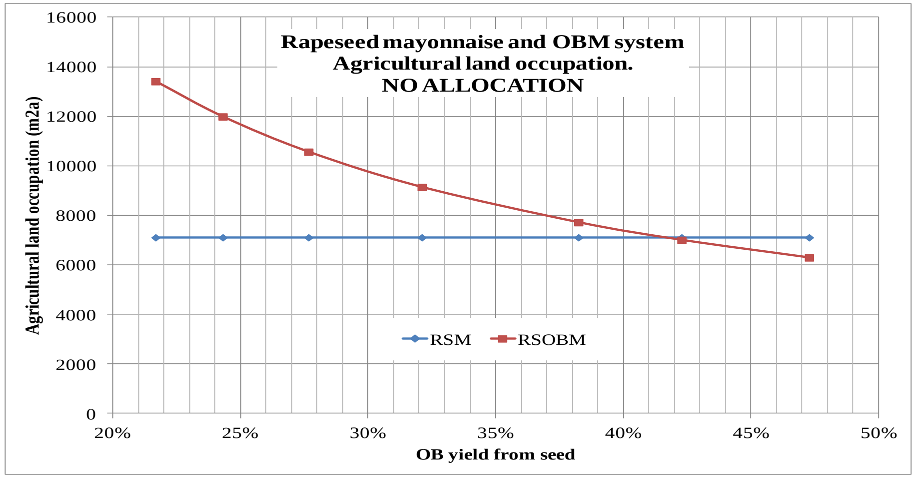


Figure F 8-8: Yield comparison, RSOBM, No allocation – ALO.

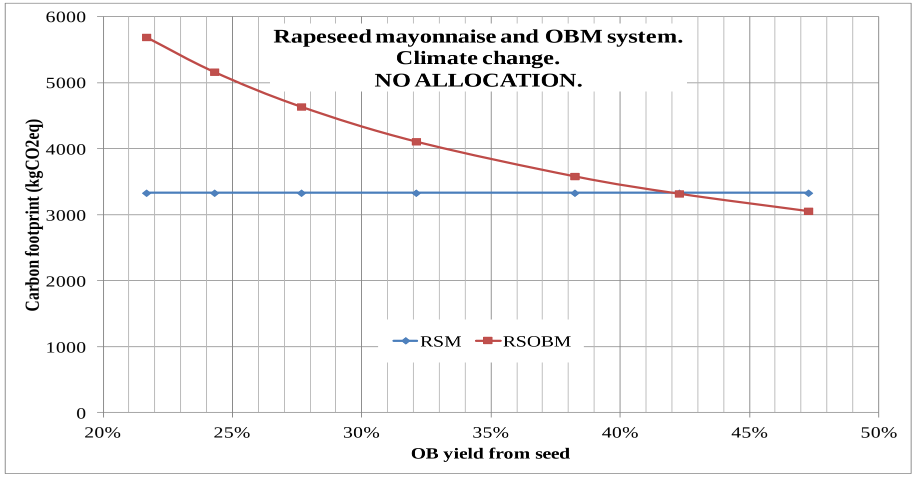


Figure F 8-10: Yield comparison, RSOBM, No allocation - CC.

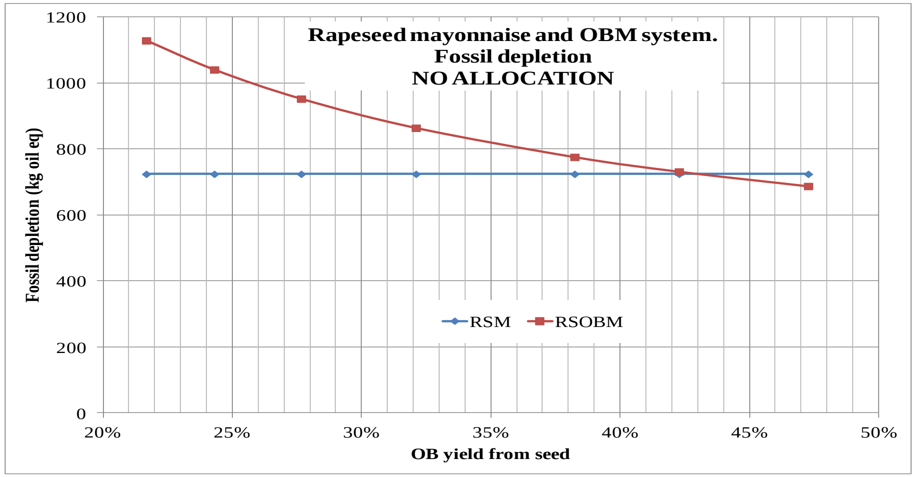


Figure F 8-9: Yield comparison, RSOBM, No allocation - FD.

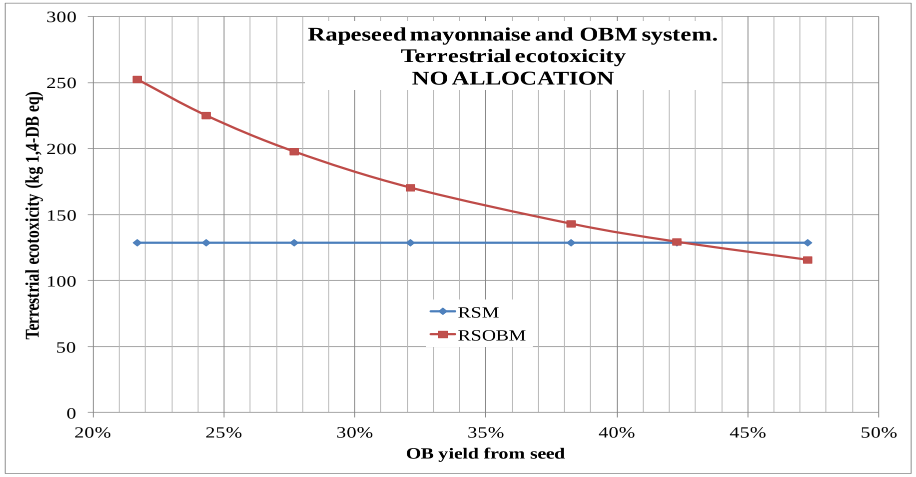


Figure F 8-11: Yield comparison, RSOBM, No allocation – TET.

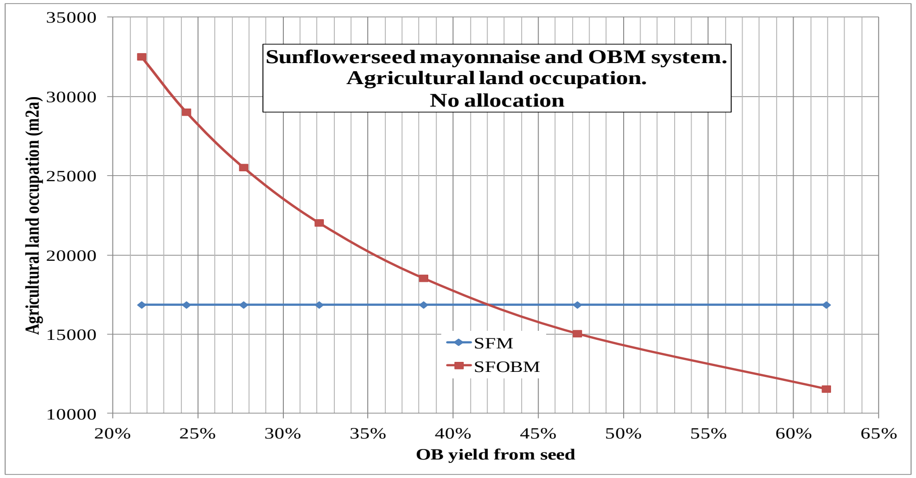


Figure F 8-12: Yield comparison, SFOBM, No allocation - ALO.

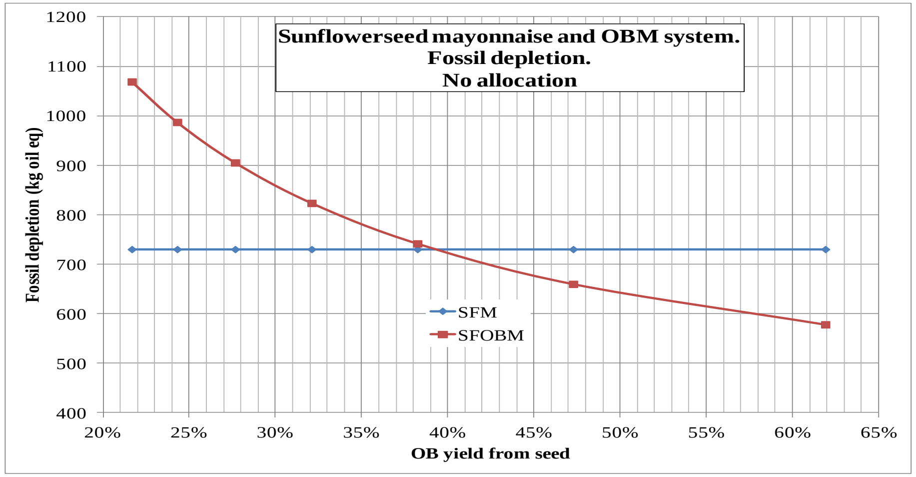


Figure F 8-13: Yield comparison, SFOBM, No allocation - FD.

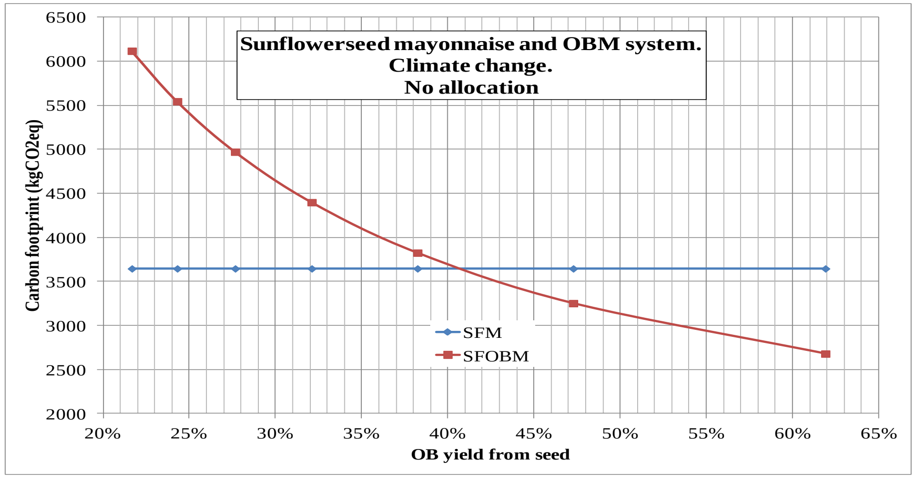


Figure F 8-14: Yield comparison, SFOBM, No allocation - CC.

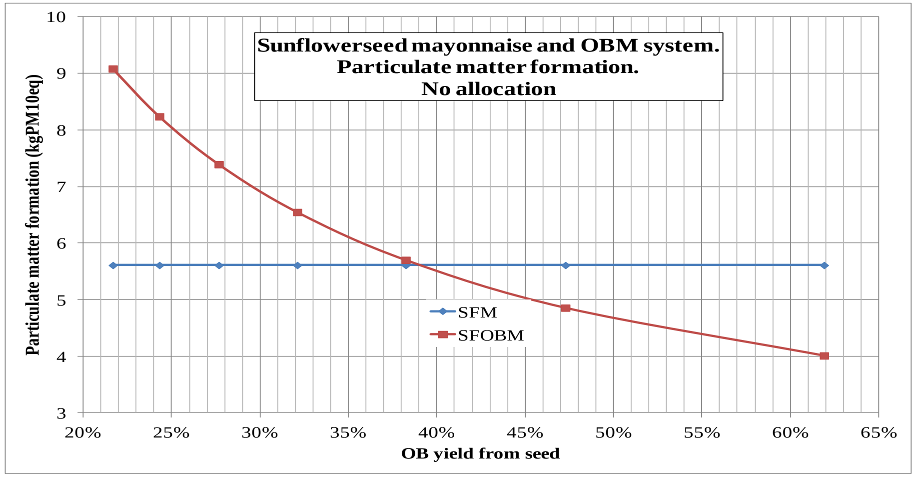


Figure F 8-15: Yield comparison, SFOBM, No allocation - PMF.

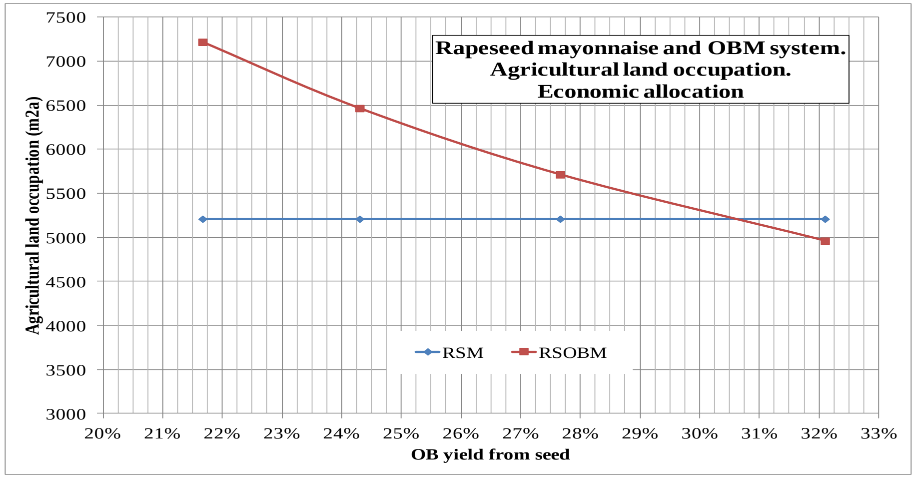


Figure F 8-16: Yield comparison, RSOBM, Economic allocation - ALO.

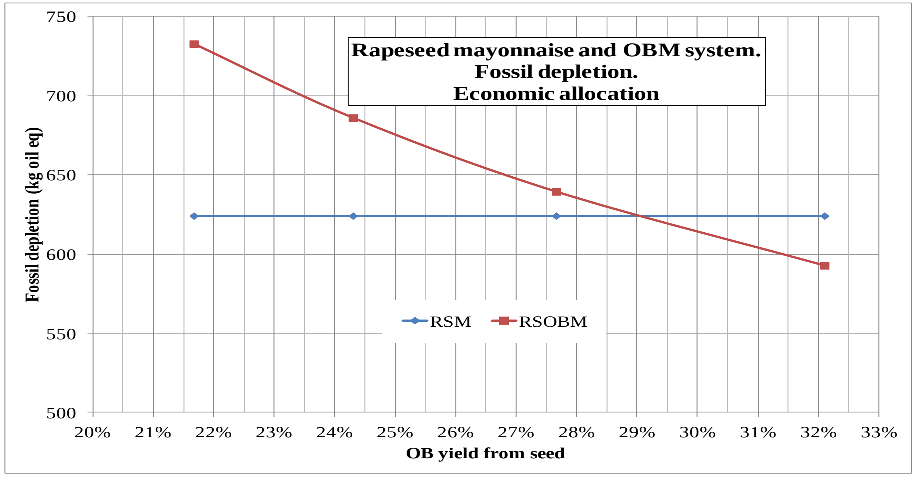


Figure F 8-8-17: Yield comparison, RSOBM, Economic allocation - FD.

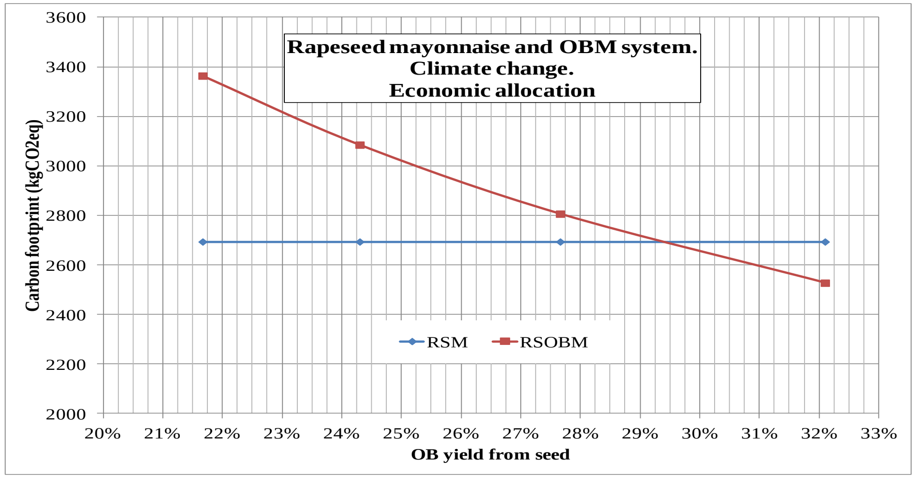


Figure F 8-18: Yield comparison, RSOBM, Economic allocation - CC.

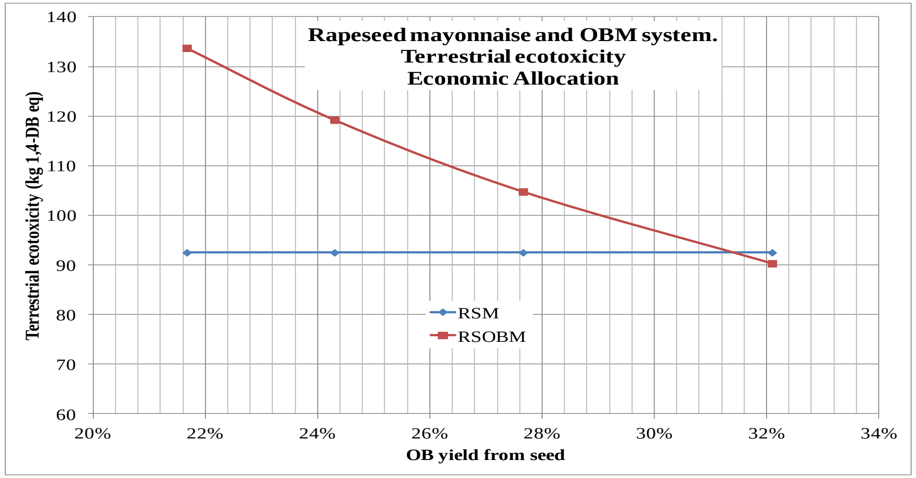


Figure F 8-19: Yield comparison, RSOBM, Economic allocation - TET.

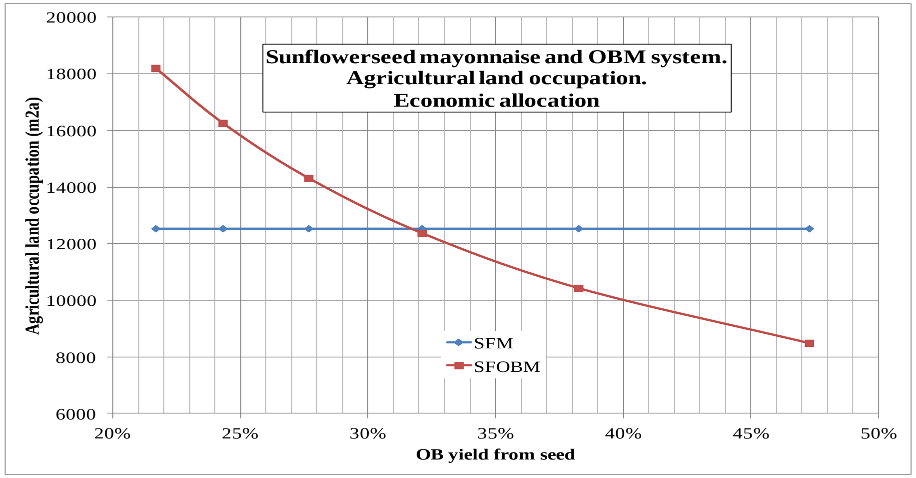


Figure F 8-20: Yield comparison, SFOBM, Economic allocation - ALO.

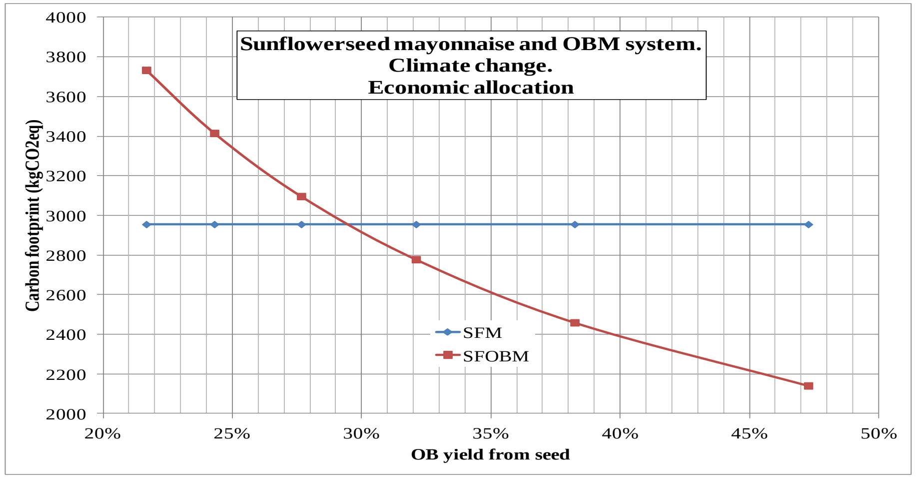


Figure F 8-22: Yield comparison, SFOBM, Economic allocation - CC.

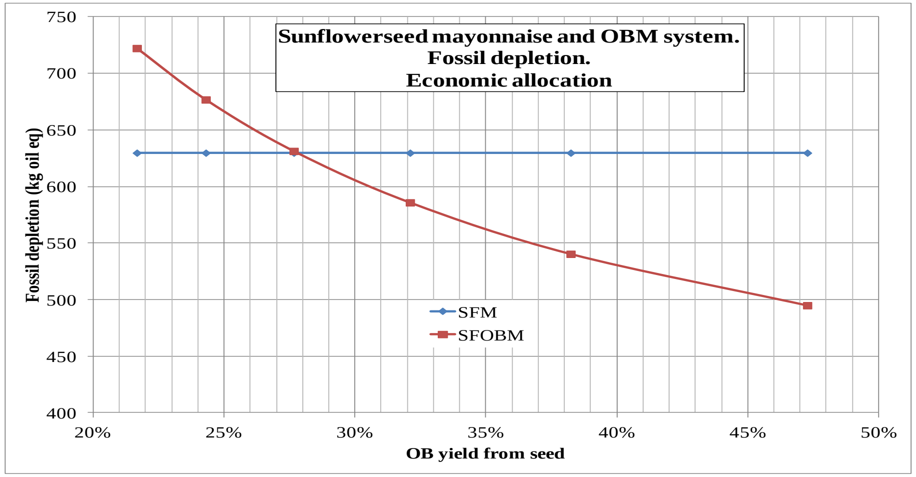


Figure F 8-21: Yield comparison, SFOBM, Economic allocation - FD.

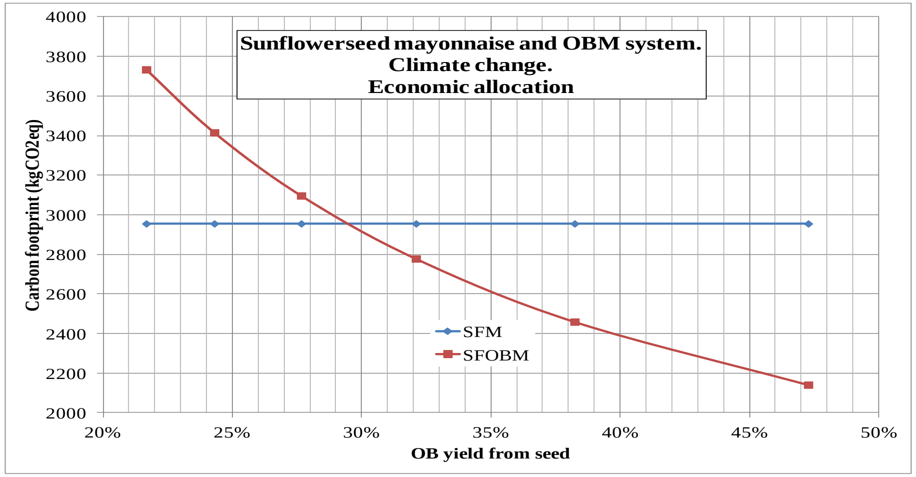


Figure F 8-23: Yield comparison, SFOBM, Economic allocation - PMF.

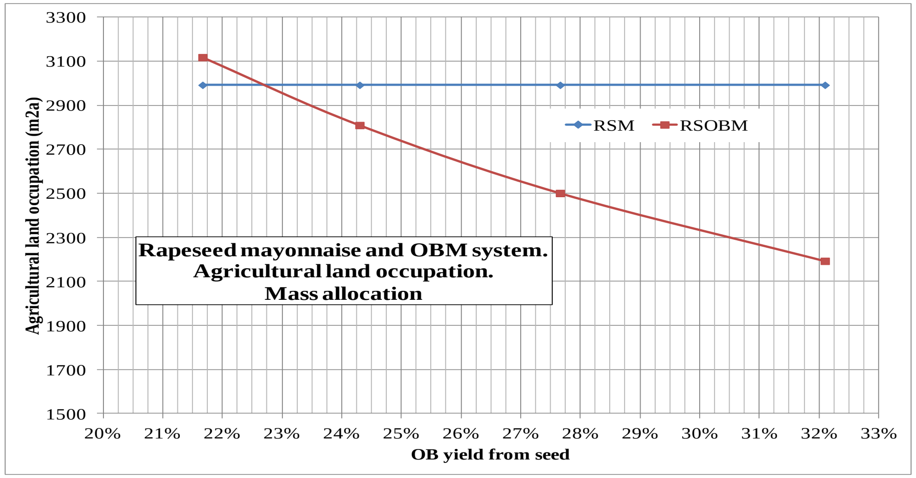


Figure F8-24: Yield comparison RSOBM, Mass allocation - ALO.

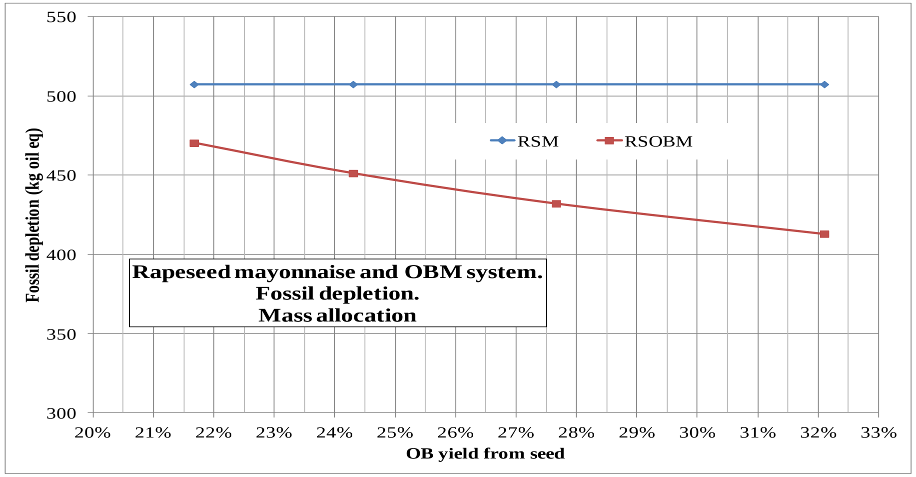


Figure F8-25: Yield comparison RSOBM, Mass allocation - FD.

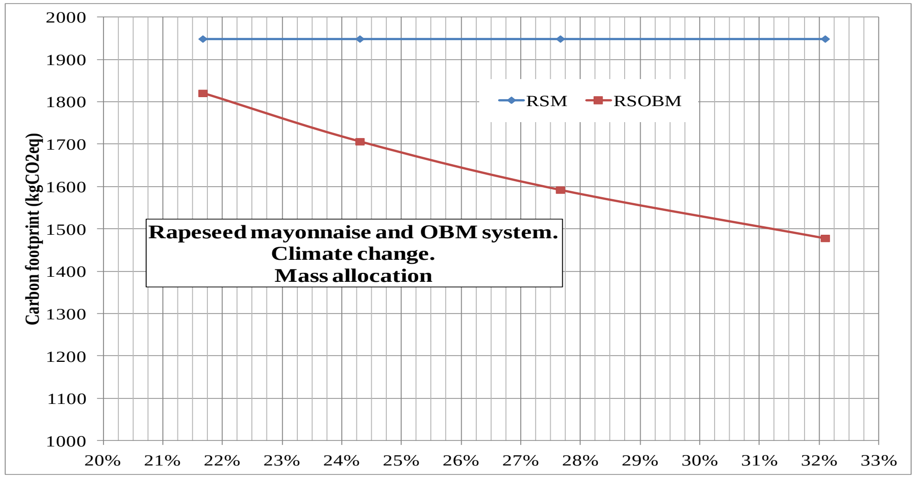


Figure F8-26: Yield comparison RSOBM, Mass allocation - CC.

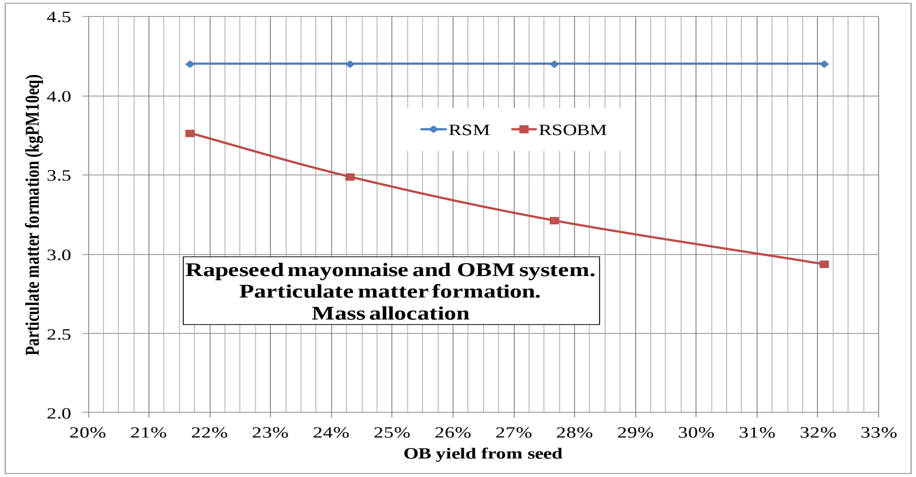


Figure F8-27: Yield comparison RSOBM, Mass allocation - PMF.

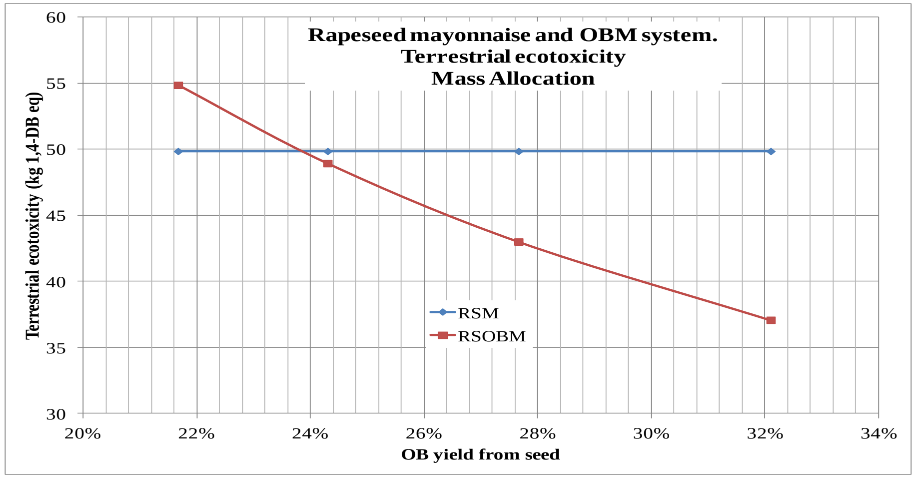


Figure F8-28: Yield comparison, RSOBM, Mass allocation - TET.

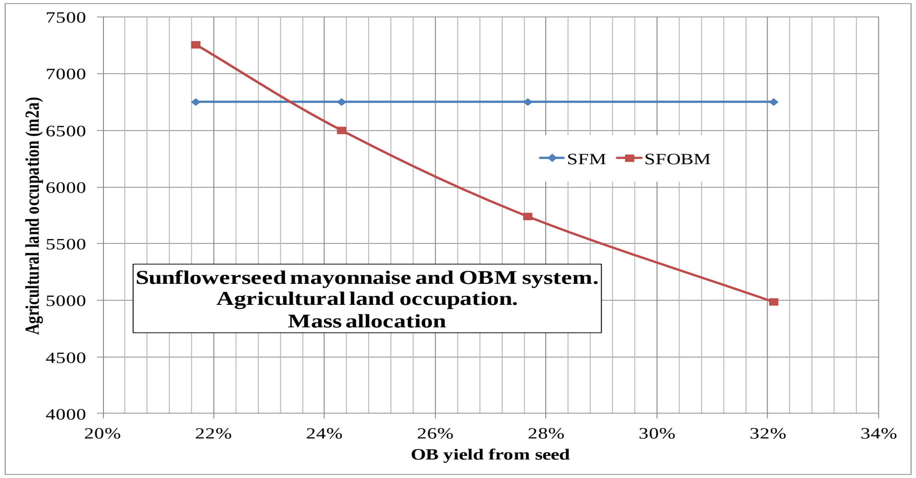


Figure F 8-29: Yield comparison, SFOBM, Mass allocation - ALO.

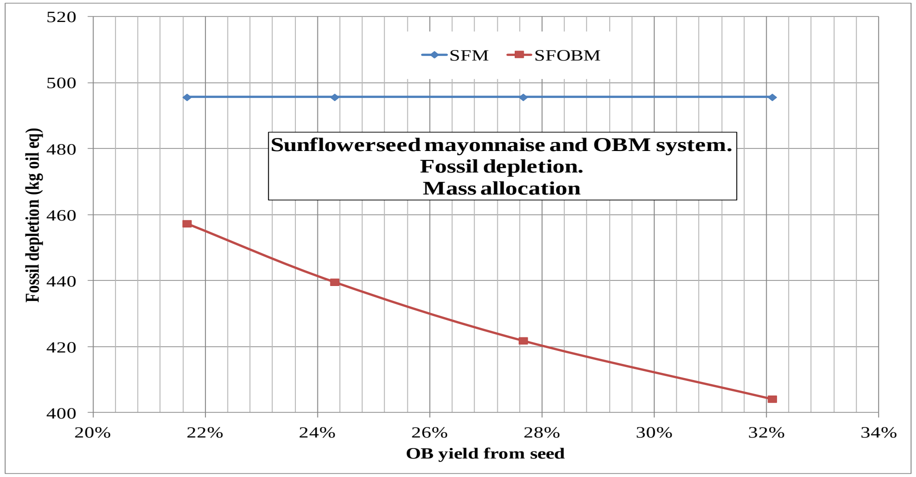


Figure F8-30: Yield comparison, SFOBM, Mass allocation - FD.

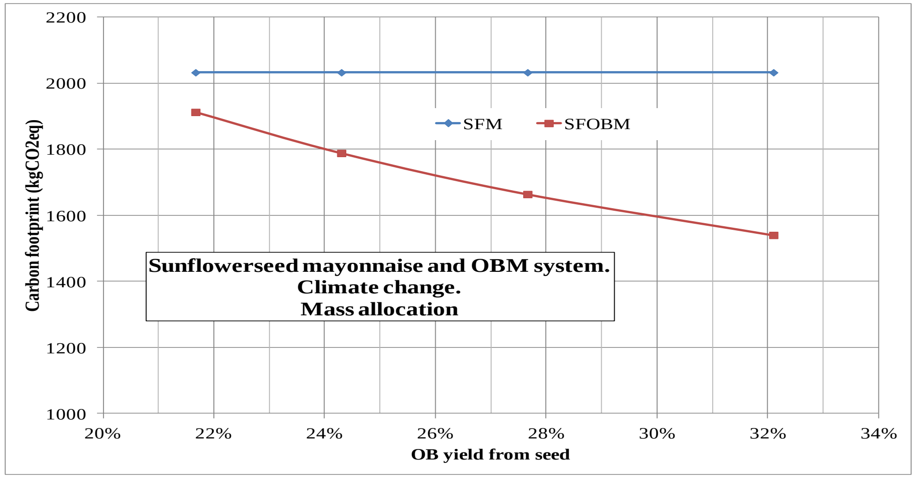


Figure F8-31: Yield comparison, SFOBM, Mass allocation - CC.

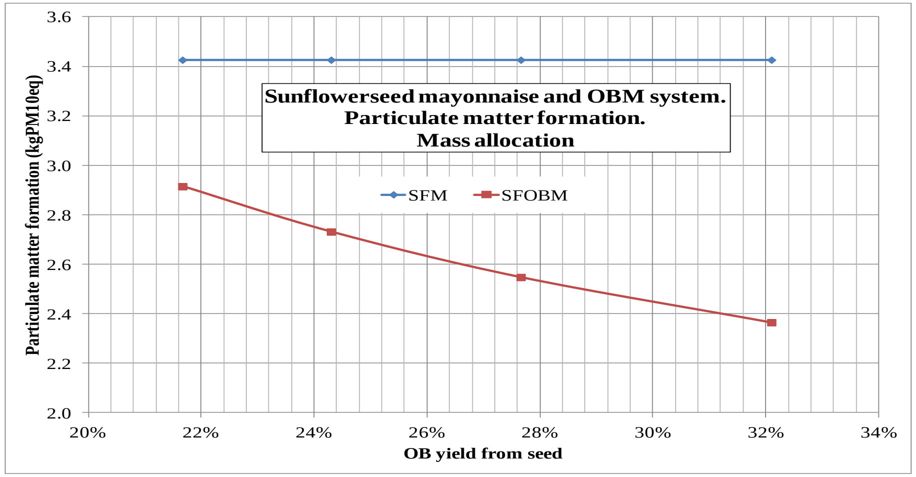
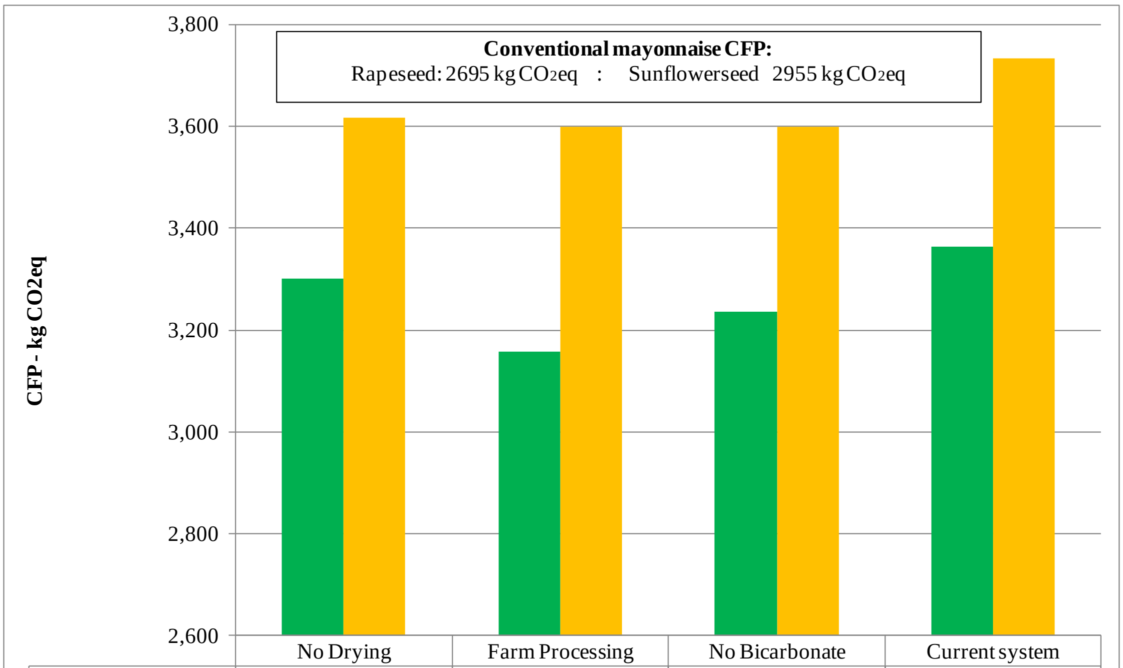


Figure F8-32: Yield comparison, SFOBM, Mass allocation - PMF.

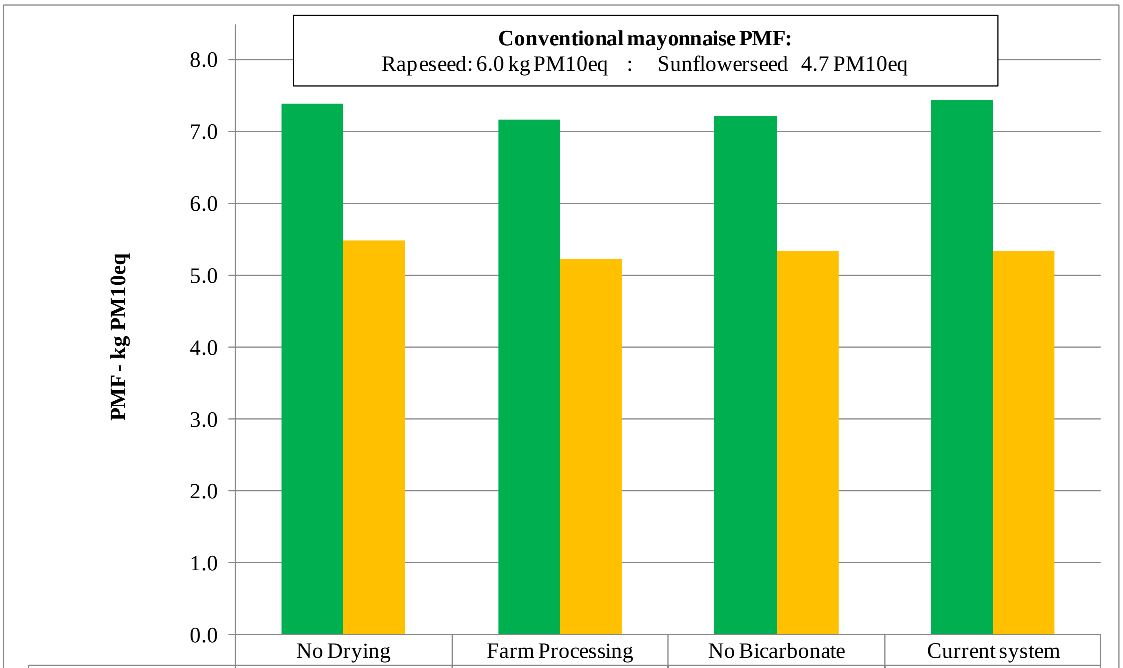
Impact category	Unit	Result with PLUS 10% meal value	Change from base	Base case	Change from base	Result with MINUS 10% meal value	Rapeseed mayonnaise
Climate change	kg CO2 eq	3247.2	3.5%	3363.8	-3.9%	3494.8	2693.7
Ozone depletion	kg CFC-11 eq	0.0003	3.2%	0.0003	-3.5%	0.0003	0.0003
Human toxicity	kg 1,4-DB eq	583.7	3.3%	603.6	-3.7%	625.9	451.7
Photochemical oxidant formation	kg NMVOC	10.9	3.4%	11.2	-3.8%	11.7	9.6
Particulate matter formation	kg PM10 eq	7.2	3.7%	7.4	-4.2%	7.8	6.0
Ionising radiation	kg U235 eq	295.1	3.2%	305.0	-3.6%	316.1	244.7
Terrestrial acidification	kg SO2 eq	33.7	4.0%	35.1	-4.5%	36.6	29.5
Freshwater eutrophication	kg P eq	0.9	3.9%	1.0	-4.4%	1.0	0.7
Marine eutrophication	kg N eq	29.2	4.4%	30.6	-5.0%	32.1	23.1
Terrestrial ecotoxicity	kg 1,4-DB eq	127.8	4.5%	133.7	-5.0%	140.4	92.5
Freshwater ecotoxicity	kg 1,4-DB eq	37.7	4.2%	39.4	-4.7%	41.2	27.6
Marine ecotoxicity	kg 1,4-DB eq	11.6	3.5%	12.1	-3.9%	12.5	8.9
Agricultural land occupation	m2a	6909.7	4.3%	7219.4	-4.8%	7567.2	5209.7
Urban land occupation	m2a	70.2	4.1%	73.2	-4.6%	76.6	51.9
Natural land transformation	m2	0.6	2.7%	0.6	-3.0%	0.6	0.5
Water depletion	m3	23.3	3.7%	24.2	-4.1%	25.2	17.5
Metal depletion	kg Fe eq	136.6	3.9%	142.1	-4.4%	148.3	96.7
Fossil depletion	kg oil eq	713.0	2.7%	732.8	-3.0%	755.1	624.2

Table 8-7: Sensitivity analysis on economic allocation using +/- 10% economic value for aqueous extraction residue.



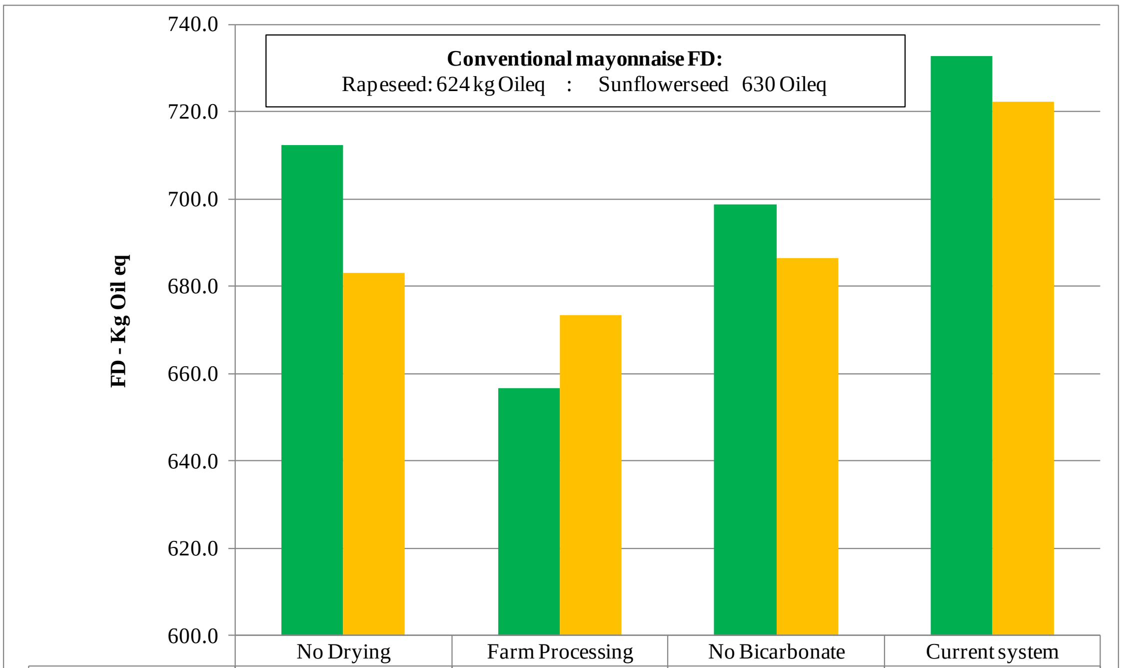
	No Drying	Farm Processing	No Bicarbonate	Current system
Rapeseed OBM	3302	3157	3236	3364
Sunflowerseed OBM	3616	3599	3599	3734

Figure F8-33: Impact of modifying process elements on CFP.



	No Drying	Farm Processing	No Bicarbonate	Current system
Rapeseed OBM	7.4	7.2	7.2	7.4
Sunflowerseed OBM	5.5	5.2	5.4	5.4

Figure F8-34: Impact of modifying process elements on PMF.



	No Drying	Farm Processing	No Bicarbonate	Current system
Rapeseed OBM	712	657	699	733
Sunflowerseed OBM	683	673	687	722

Figure F 8-35: Impact of modifying process elements on FD.